EFFECTS OF WOODY VEGETATION REMOVAL ON SOIL WATER DYNAMICS

IN A SOUTH TEXAS SHRUBLAND

A Thesis

by

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ABSTRACT

Ecosystem changes from grassland to shrubland in the Rio Grande Plains are thought to have negative effects on the hydrology of the region. The increase in woody plants, known as woody encroachment, may alter the amount of water moving beyond the root zone of plants. Water moving beyond the root zone is referred to as deep drainage, and has potential to become aquifer recharge. A vegetation manipulation project was designed to understand the effects of woody vegetation removal on soil water dynamics in the recharge zone of the Carrizo-Wilcox aquifer of south Texas. The primary objective of the project was to determine the potential to increase groundwater recharge through woody vegetation removal. To understand the effects of vegetation removal on various soil textures we studied changes in soil water, rooting depth, and the role of water redistribution by woody vegetation. Woody vegetation was removed using common methods of cut-stump and roller chop across three soil types. Soil water contents and changes were measured using neutron moisture meter to a depth of 180 cm. Average rooting depth was determined across three soil types. Soil and stem water stable isotopes were used to understand soil water movement.

Rooting depth was determined to between 140 and 160 cm for all soil textures. Soil water content and changes were analyzed at three depth increments: 0-60, 60-120 and 120-180 cm. ANOVA analysis showed that there was no treatment response in average soil profile water in the sandy or sandy loam soils. There was a significant decrease in soil profile water for clay loam soil in response to roller chopping. Changes

in soil profile water were the greatest in the sandy roller chopped soils. Below 120 cm, three months had significant differences in change in soil water in the sandy roller chop plot. During dry conditions, Honey mesquite shifts water use to deeper in the soil profile. In clay loam soils under dry conditions there is evidence of water being moved up from below 2 m soil depth to drier shallow soils. Roller chopping in sandy soils is the vegetation removal treatment and soil type most likely to result in water moving beyond the root zone. Although treatments had significant effects on soil moisture dynamics that interacted with soil type, we did not find support for deep drainage effects over the Carrizo-Wilcox aquifer from woody vegetation removal.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER I

INTRODUCTION AND LITERATURE REVIEW

Introduction

Water is a resource that is undeniably important. Ecosystems, people and wildlife all depend on fresh water for life. Freshwater exists generally as surface water or groundwater. In areas where precipitation is low, groundwater is often heavily relied on for human consumption as drinking water and irrigation for food crops. Rangeland ecosystems rely on rainfall, and changes to plant distributions and compositions can alter the balance between above ground and below ground water supplies.

In south Texas, ground water is relied on heavily. One of the major sources is the Carrizo-Wilcox sand aquifer. This is an unconsolidated sand aquifer that is thought to recharge through stream beds and diffuse recharge through the soil. The vegetation within the recharge zone is that of the Tamaulipan biotic province, characterized by thorny shrubland vegetation, and dominated by honey Mesquite. Generally, this area is described as the Rio Grande Plains ecoregion. The area of focus for this research is in the northern Rio Grande Plains, where the Carrizo-Wilcox formation outcrops and recharge to the aquifer is thought to occur.

During the last century grasslands throughout the world have experienced increases in woody vegetation. This process is known as woody encroachment. Woody encroachment is the increase in density, cover and biomass of native woody plants into primarily arid and semiarid grasslands (Van Auken 2000). There are various factors such as grazing, fire suppression, changes in rainfall, temperature and CO_2 levels that are causing encroachment, however, most of the recent changes seem to be directly or indirectly caused by anthropogenic factors or management decisions (Van Auken 2009). The extent of woody encroachment is not limited to any specific region of the world. Woody plants have increased in many grasslands and savannas in Africa, Asia, Australia, South and North America (Boutton et al. 1998).

Woody encroachment throughout the world has been heavily discussed and debated. The usual reasons cited are those mentioned above, as well as global climate change, chronic high levels of herbivory, seed dispersal, changes in herbaceous competition and changes in small animal populations (Van Auken 2009). The changes in woody density occurring in the southwestern grasslands of North America have occurred in last 160 years, and are most strongly associated with the introduction of cattle and cattle grazing systems (Archer 1995).

Grazing begins the process of woody encroachment by reducing herbaceous cover. The shift from grassland to shrubland is initiated by a reduction in herbaceous cover. Once the process is initiated, these landscapes show a continued increase in woody vegetation and a continued decrease in herbaceous cover leading toward a closed canopy wooded ecosystem (Walker et al. 1981). This process which decreases grass root biomass and depth causes the replacement of long-lived deep rooted grasses by short-lived shallow rooted grasses (Boutton et al. 1998). The increase in woody plants also alters the distribution of biomass form predominantly below ground to predominantly above ground (McKinley et al. 2008).

Mesquite has been shown to be an important facilitator of increases in woody plants following reduction in herbaceous plants by grazing (Archer et al. 1988). Woody clusters are initiated by the colonization of honey mesquite which facilitates the recruitment of other woody plants beneath its canopy (Boutton et al. 1998). The development of woody assemblages fits the facilitation model of Connel and Slatyer (1977) (Archer et al. 1988). The facilitation model states that early successional species modify the environment to more suitable condition for later-successional species to establish and proliferate (Connell and Slatyer 1977). We have seen evidence for this model in the Rio Grande Plains as woody encroachment has changed the landscape and is moving towards a monophasic woodland (Archer 1995).

Ecosystem changes in the Rio Grande Plains have recently led to concerns about how native vegetation may be altering the water budget. Areas formerly dominated by grasslands, such as those of the Rio Grande Plains, are important from food production, economic and ecological perspective (Briggs et al. 2005). These factors contribute to the need for research on interactions of water and woody vegetation on the Carrizo-Wilcox aquifer.

Current demands on the Southern portion of the Carrizo-Wilcox aquifer for water are primarily irrigation and municipal supply which accounts for 90% of groundwater extractions (www.twdb.state.tx.us). There is political pressure to limit construction of new surface reservoirs (Olenick et al. 2005). This limitation on new surface reservoirs has increased demands on aquifers and has led management agencies to find alternative means to meet the needs of the people and environment. The increase in woody plants in grasslands has also led to an inability of grasslands to support the large population of grazers and the pastoral economy of the past (Reynolds et al. 2007). Concerns about mitigating economic and hydrologic impacts of woody encroachment are incentivized by a general preference for grassland over dense brush cover by landowners (Olenick et al. 2005).

Purpose

Not all water that enters the soil is lost by evaporation or transpiration. Water that moves beyond the root zone is no longer able to be accessed by plants. This water is often referred to as potential recharge or deep drainage. This water can stay in the unsaturated zone or continue to move to the groundwater table (Healy 2010). Based on the two-layer hypothesis, increases in groundwater yield are theoretically feasible through removal of deep rooted species (Carlson et al. 1990). To test the validity of this theory in the Rio Grande Plains, we have chosen to conduct research in the recharge zone of the Carrizo-Wilcox aquifer. The recharge zone is the area where water moving beyond the root zone is believed to migrate towards the underlying aquifer.

The purpose of the research is to determine the potential to increase groundwater recharge through the removal of woody vegetation. The main objective of this project is to determine if there is water moving beyond the root zone across a variety of soil textures in response to common woody vegetation removal methods. This determination will be made through measurement and analysis of soil water, plant root biomass distributions, and soil and stem water stable isotopes.

Literature review

Woody encroachment in south Texas has various impacts on the region. The shift from grassland to woody shrubland has been met unfavorably by local landowners. There is a desire to manage the shrubland vegetation to maximize water yield while meeting the needs of the various stakeholders. In the following section I will provide background information on the current understanding of the issue. The information will focus on research that has been conducted using methods similar to those of this study and research from similar regions.

The following literature review will describe the effects of woody encroachment and the resulting effects on aquifer recharge. There will be an emphasis on the importance of root distribution within the soil profile and how those differ across regions. I will discuss the use of stable isotopes as a tool to understand soil water dynamics. I will then discuss the roles of vegetation and soil texture and how those influence soil water dynamics.

Impacts of woody encroachment

Under certain lithologies and climates the reduction of woody plant cover can enhance water yields (Dugas and Mayeux 1991). A shift from shrubs or forests to grasslands is believed to increase groundwater recharge based on a reduction in rooting depth and plant cover (Moore et al. 2010). Changes in groundwater recharge as a result of land use/ land cover change have been documented in the Pampas of South America (Joggaby and Jackson 2004), in multiple locations across Australia (Petheram et al. 2002) and the High Plains of Texas (Scanlon et al. 2005). In an attempt to develop a generic relationship that could be broadly applied to predict potential for changes in groundwater recharge, Petheram et al. (2002) emphasized the effect of soil type on such estimates.

Numerous studies on the Edwards Plateau have been conducted on the role of vegetation and its influences on potential recharge (Wilcox et al. 2006b). This research, although conducted in close proximity to the Carrizo-Wilcox aquifer does not translate because of differences in vegetation and underlying lithology. The dominant vegetation of the Edwards plateau is Ashe juniper and is underlain by the Edwards aquifer (Bailey 1995). The Edwards aquifer is a karst aquifer and karst aquifers have complex and original characteristics such as large voids, high flow velocities and high flow rate springs that make their hydrologic function different from other types of aquifers (Bakalowicz 2005). The Rio Grande Plains is primarily underlain by the Carrizo-Wilcox aquifer which is an unconsolidated sand aquifer. So, in spite of the abundance of research conducted nearby, the findings related to recharge to the Edwards aquifer do not necessarily translate to the Rio Grande Plains and the Carrizo-Wilcox aquifer. Beyond Texas there are areas with large sedimentary basins like the Gulf Coast coastal plain to which these findings might be applied (Fogg 1986). The understanding of the Carrizo-Wilcox is not as extensive as some other aquifer systems in Texas, but the findings could be broadly applied to similar climatic and vegetative regimes, especially where woody encroachment is of concern.

Woody encroachment is often associated with ecosystem degradation, declines in forage productivity, declines in biodiversity and socioeconomic potential as well as increased erosion (Huxman et al. 2005). Many of these plants are considered "woody weeds" in S. Africa and Australia, and act at the exclusion of herbaceous vegetation (Hobbs and Mooney 1986). In N. America, Baccharis invasion into grasslands in California has been abrupt in time and space and resulted in declines in herbaceous vegetation (Hobbs and Mooney 1986). There has also been invasion of grasslands by Artemisia in the Sierra Nevada mountains of California (Darrouzet-Nardi et al. 2006), and Larrea into the grasslands of the Southwest (Grover and Musick 1990).

The invasion by woody plants into grasslands is also occurring in the plains of S. Texas. What was once grassland is now dominated by subtropical thorn woodland complex of dense thickets of shrubs and small trees (Boutton et al. 1998). Multiple lines of evidence show the expansion of woody plants in S. Texas has occurred over the last 50-100 years (Boutton et al. 1998). Reports of increased density of woody plants in arid and semiarid grasslands of southwestern North America date back to the late 1800's (Van Auken 2009).

Regional vegetation

In south Texas where this research will be conducted the potential vegetation of southern Texas and northern Mexico has been classified as Prosopis Acacia savanna (Kuchler 1964). The area has been broadly described by Bailey (1995) as a semiarid steppe, which is the transition between the deserts of the southwest and the more humid surroundings, with soils in the orders of Alfisol Mollisol and Vertisol. Bailey defines this areas as one where from May to October, potential evaporation is about twice the precipitation (Bailey, 1995). More recently and in greater detail, the landscape has been classified by Brown (1998) as Tamaulipan thornscrub biotic community, characterized by particular species of plants. The Tamaulipan thornscrub covers 188 km^2 (Brown 1998), and is dominated by dense woodlands, while some landscapes remain as savannas, while still others are thought to be developing into closed-canopy woodlands (Brown and Archer 1990). The variability of soil characteristics interact with rooting patterns and result in a varied abundance and distribution of life-forms across these landscapes (Brown and Archer 1990). The most abundant life forms are medium and small shrubs (Návar et al. 2004). The dominant shrub species of this ecosystem are *Acacia berlandieri* Benth., *A. farnesiana* (L.) Wild., *A. rigidula* Benth., *Calliandra conferta* Gray, *Celtis pallida* Torr., *Condalia hookeri* M.C. Johnst., *Cordia boissieri*

DC., *Diospyros palmeri* Scheele, *Diospyros texana* Scheele, *Ehretia anacua* (Terán & Berl.) I.M. Johnst., *Eysenhardtia polystachya* (Ort.) Sarg., *Eysenhardtia texana* Scheele, *Forestieria angustifolia* Torr., *Fraxinus greggii* A. Gray, *Gochnatia hypoleuca* DC., *Helietta parvifolia* (Gray) Benth., *Leucophyllum texanum*, *Malpighia glabra* L., *Mimosa biuncifera*, *Pithecellobium pallens, Pithecellobium ebano, Prosopis laevigata, Prosopis glandulosa, Schaefferia cuneifolia*, and *Zanthoxylum fagara* (Návar et al. 2004).

<u>Rooting patterns</u>

Root distributions are important to understanding groundwater fluxes, soil litter decomposition, carbon sequestration and nutrient cycling (Jackson et al. 1996). In the shrublands of S. Texas, we know some species can have very deep roots, but the common habit of species across the environment is poorly understood (Boutton et al. 1998). The implications for woody plant encroachment for both water and biogeochemical cycles are poorly understood (Huxman et al. 2005). Woody plant removal has the potential to decrease overall rooting depth in the soil which may affect potential groundwater recharge. For this reason it is important to quantify rooting depth in the recharge zone of the Carrizo-Wilcox because this is highly variable across species composition, soil type and space (Boutton et al. 1998).

Rooting depth can vary greatly with soil texture and with species composition. Root distributions in a fine textured subsoil are dominant in the top 1.5 m in the mesquite woodland of the Rio Grande Plains, but in deep coarse-textured soil plants utilize water from deeper than 1.5 m (Canadell et al. 1996). In fine-textured soils, water should remain in the upper horizons and favor grasses by limiting the downward movement of water to lower horizons (Brown and Archer 1990). Conversely, coarse-textured soils permit greater infiltration and deeper percolation of water which should favor woody plants exploiting water at greater depths (Brown and Archer 1990).

Interestingly, most of the rooting distribution data of savannas shows that trees have the majority of their roots in the topsoil, this is probably because the near surface is where moisture and nutrients co-occur (Ludwig 2004). Shallow root distributions in coarse grained soils could also be explained by considering small precipitation events. Small rain events account for a relatively large proportion of precipitation and have a large contribution to the primary productivity of semi-arid grasslands (Sala and Lauenroth 1982). Root biomass distributions in the eastern Rio Grande Plains were found to decrease exponentially with depth in groves and linearly in grasslands (Midwood et al. 1998). In the same study, roots proliferated above argillic horizons and in areas with a clay-pan stem water closely matched soil water from the argillic horizon (Midwood et al. 1998). In areas where crack occur in dry clay soils, mesquite can grow very deep roots (Huxman et al. 2005). Moore et al. (2010) found higher bulk densities to be associated with deeper rooting depths, with surprisingly no relationship between rooting depth and texture. This brief synthesis of rooting depth distributions highlights the variability and plasticity of roots in relationship to root-density and soil-texture interactions (Canadell et al. 1996, Midwood et al. 1998).

There has been research that shows that the Tamaulipan plains ecosystem following woody encroachment generally conforms to the 2-layer hypothesis (Ansley et al. 2007). The 2- layer hypothesis states that grasses typically are more shallowly rooted and shrubs are typically more deeply rooted, and therefore avoid competition for water. Although this is a general assumption regarding root distributions, it does not fully describe below ground root distributions. There are cases where single plants have roots been found to extend very deep into cracks in rocks. There has also been research that examines the general rooting habit of plants in greenhouse experiments. Such examples don't necessarily represent the general habit of roots in a natural system. We know the tendency of some species to have very deep roots at some sites, but the common habit of some of the deep rooting species is unknown (Boutton et al. 1998).

In the Rio Grande Plains of south Texas, there has been some research done to understand rooting depth and root distribution of woody plants, and specifically *Prosopis glandulosa*. Of the work that has been done, results are not necessarily in agreement with each other. Jackson et al. (1996) cites varying results in their synthesis paper: Heitschmidt et al. (1988) found 90% of roots in the top 133 cm, Watts (1993) found 92 % above 120 cm and Canadell et al. (1996) claims 30% of root biomass exists below 1 m. Jackson et al. (1996) found only two studies have quantified root depths to 2 m for *Prosopis glandulosa*, a common species of the Rio Grande Plains known to be a deep-rooted woody plant. Of the two studies, one was a root excavation where 1 mature tree and 12 immature trees were studied (Heitschmidt et al. 1988). The excavation ended at 1.3 m for the mature tree and found the majority of lateral roots in the top 30 cm. The root distributions for immature trees found 42% of roots above 30 cm, 81% within the top 1 m and only 4 % of the roots at 2 m. In the second of the two studies, Midwood et al. (1998) found 84% of root biomass for grove clusters (an amalgam of woody plants, including a *Prosopis glandulosa*) above 60 cm and root biomass decreased exponentially to a depth of 2 m. Interestingly, stable isotope analysis from Midwood et al. (1998) showed water acquisition from below 1.5 m in spite of 84% of the root biomass being above 60 cm. Stable isotope data may validate claims that *Prosopis glandulosa* is a facultative phreatophyte (Heitschmidt et al. 1988) or even that single roots can reach depths of 50 m (Phillips 1963).

Two areas similar to the Rio Grande Plains that have experienced encroachment by P*rosopis* and other woody species are the Rolling Plains ecological region and the Edwards Plateau. These areas are not analogous to the Rio Grande Plains due to differences in substrate and climate. They do however offer some insight to rooting depth and thus the effect of woody encroachment on soil water dynamics.

A study in the northern Rolling Plains ecological area on *Prosopis* assumed that because no soil moisture was found below 2 m then there was no root biomass (Ansley et al. 2007), yet this assumption disregards preferential flow paths and potential for roots to follow those to deeper water. The same study found support for the two-layer hypothesis in the *Prosopis* ecosystem and that mesquite roots were concentrated at 10-90 cm. On the nearby Edwards Plateau, mesquite roots extended to depths of 1.5-2.5 m where growth was inhibited by unfractured limestone bedrock (Eggemeyer and Schwinning 2009). These authors suggested the roots would have grown deeper in the absence of growth inhibiting bedrock. Stable isotope data from Eggemeyer and Schwinning (2009) found differences in depth of acquisition across time and species, but not among different sizes of single species.

<u>Stable isotope applications</u>

Stable isotopes have been used extensively to show depth of water acquisition by single species as well as partitioning of water by coexisting species. Darrouzet-Nardi et al. (2006) showed sagebrush in a California meadow acquired 10-30% of its water from sources < 30cm. Using stable isotopes Dodd et al. (1998) showed that Atriplex shrubs mainly used water from spring and summer precipitation events and that water utilized was extracted from subsurface layers. Support for the 2-layer hypothesis was also shown by stable isotope uses by in shortgrass steppe of northeastern Colorado (Dodd et al. 1998), and in semi-arid grasslands of southeastern Arizona (Weltzin and McPherson 1997). Evidence that trees and grasses do not always have complete niche separation was shown by Ludwig et al. (2004) in an east African savanna. In cases where a small fraction of roots are tapping water from the water table, the amount of water transferred into the plant may be large (Canadell et al. 1996). Alternatively, where it seems that a plant may be accessing stream water they may be utilizing surface water as in the case shown by Ehleringer & Dawson (1996). Use of stable isotopes by the above researchers has shown that we can quantitatively assess the spatial and temporal use of soil water by different plants.

Plant and soil attributes have large influences on soil water and manipulation of these components can have influence on the soil-water balance (Moore et al. 2010). Stable isotopes are a valuable tool for understanding the soil water pathways and can be used to differentiate loss by evaporation or transpiration (Allison et al. 1983). The general form of a soil profile undergoing loss by evaporation is one that increases from a low isotopic value at the surface to a maximum isotopic value at some depth (Allison et al. 1983). The water at a vegetated site is expected to be depleted of heavier isotopes relative to a non-vegetated site due to lower soil water contents. The effects of evaporation and loss of lighter isotopes on soil water isotope values are thus exaggerated compared to a non-vegetated site. The end result is a slightly lower slope and a more positive y-intercept value for vegetated vs. non-vegetated sites (Allison et al. 1983).

Stable isotopes can help differentiate between losses of water by evaporation (a fractionating process), and transpiration (a non-fractionating process) by the slope of the relationship of $\delta^{18}O$ and δD (Allison et al. 1983). The depth of influence of evaporation on a soil can also be estimated by comparing soil moistures to field capacities (Komor and Emerson 1994). Based on the field capacity method, in the sand plains in the upper Midwest, it was found that the evaporation front did not extend below 30 cm. This was supported by the increase in carbonate concentrations indicating a long term barrier to leaching (Komor and Emerson 1994). Research in a Brazilian cerrado showed relatively constant δD values at 50 cm depth during the prolonged dry season indicating no evaporation (Jackson et al. 1999). In a controlled laboratory experiment where soils were left to undergo evaporation, after 1 year, evaporation did not extend below a depth of 30 cm (Allison et al. 1983).

The isotopic composition of a soil profile as a result of percolation of water through heterogeneous, variably saturated soils is a smoothing effect with depth of seasonal differences (O'Driscoll et al. 2005). The depth where seasonal variation is lost, isotopic variation is dampened and where the isotopic profile reaches a maximum value, is known as the "critical depth" (Clark and Fritz 1997). The critical depth of soils varies based on vegetation and soil characteristics (Clark and Fritz 1997). O'Driscoll et al. (1997) found that dampening depths were the deepest in open fields, and the shallowest below the canopy of a forested area in spite of having similar soil characteristics. In the same study, clay soils were found to reduce the dampening depth. This research reported dampening depths ranging from 1.6 – 2.85 m, but also cited other work that reported seasonal variations in isotopic composition to depths of 9 m (Clark and Fritz 1997). Research conducted during the dry season in the Brazilian cerrado showed a critical depth of 4 m (Jackson et al. 1999). The critical depth in soils can vary based on timing and amount of precipitation, antecedent moisture conditions, and isotopic composition of precipitation (O'Driscoll et al. 2005).

Soil water movement

Water generally moves into soil by gravity and out of the soil by, evaporation, deep drainage, or transpiration. Some water movement within the soil profile can occur through the matrix due to potential differences or through preferential flow. Plants also have the ability to move water within the soil profile; this is commonly referred to as hydraulic lift. Hydraulic lift is the process of passive movement from wetter to drier portions of the soil profile via root systems (Jackson et al. 1999). Early work by Caldwell et al. (1998) defined a few terms new to the study of root water dynamics related to hydraulic lift. Reverse flow is the process of water moving from roots to soil. Water moving out of the roots into the soil is known as efflux (Caldwell et al. 1998). Efflux primarily occurs at night when transpiration is suppressed. The process of hydraulic lift although potentially widespread, has been reported in 19 woody taxa and 8 herbs and grasses. The process commonly occurs in semi-arid environments, but the only requirement for it to occur is occasionally dry soils (Caldwell et al. 1998).

The process of hydraulic lift is driven by differences in water potential such that physics allow for the movement of water from wetter shallow soils into deep drier soils via roots (Caldwell et al. 1998). The first reported cases of water being moved by roots to deep dry layers were in *Grevillea robusta* and *Eucalyptus camaldulensis* studied by Burgess et al. (1998). Water that was acquired from rehydrated surface soils was used for “refilling” the stem reservoir, and based on calculations, was also moving into the surrounding deeper drier soils (Burgess et al. 1998). Through two complimentary methods, Burgess et al. (1998) were able to show the reversal of hydraulic lift. They proposed hydraulic lift would more accurate and comprehensively described as “hydraulic redistribution”.

Some of the limitations for hydraulic lift to occur are related to the hydraulic conductivity and persistence of roots. For example, hydraulic lift may only occur for a portion of the year as roots progress from a state that allows, to one that does not allow hydraulic lift (Caldwell et al. 1998). Additionally, if an air gap occurs between soil and root, hydraulic lift will be inhibited (Caldwell et al. 1998). Under cloudy conditions, the effects of hydraulic lift are dampened because the daytime drying of shallow soils is one of the drivers (Caldwell et al. 1998).

Hydraulic lift was first shown to exist in the *A. saccharum* tree by Dawson (1993). He was able to show that the transfer of water from deep moist layers to dry layers was important to the performance of neighboring shallow rooted plants. The benefit of hydraulic lift to the *A. saccharum* tree is not known, but the mechanism by which hydraulic lift occurs appears to the same as that of the mesquite studied by Mooney et al. (1980) in the Atacama desert. One of the first reported cases of hydraulic lift in occurring an Acacia, or an African tree was done so by Ludwig et al. (2003) in the species *Acacia tortilis*. This Acacia species exhibited diel fluctuations which were attributed to water movement at night when the leaf stomata are closed and the major water potential gradient was between the wet deep roots and the shallow dry roots (Ludwig et al. 2003). The connection between hydraulic lift and stomates has been supported by the reverse pattern for hydraulic lift observed in CAM plants where the movement from wet to dry soil layers occurs during the daytime (Caldwell et al. 1998). The soils in which the Acacia was studied are sandy loams with very low hydraulic conductivities, which exclude the possibility of vertical or lateral movement by capillarity or mass flow based on the time frame of observation (Ludwig et al. 2003). There have been other studies that show improved soil nutrient concentration and higher floristic diversity under or very near tree canopies. Dawson (1993)suggests the improved soil condition near tree canopies and increased plant status may be attributed to hydraulic lift by some of the tree species.

Hydraulic redistribution has many implications for the maintenance of root viability and growth in dry soils as well as well as implications for ecosystem functioning and plant-plant competition or facilitation. The maintenance of an active root system allows for rapid response to small rainfall events, and this response may be an important mechanism for drought avoidance in plants growing in climates with short intense wet seasons such as Mediterranean and sub-tropical climates (Burgess et al. 1998). In the instance where shallow soil water is redistributed out of the reach of shallow rooted plants, positive plant-plant interactions such as "water parasitism" may be counteracted (Burgess et al. 1998). Additionally, this process of moving water into deeper soil layers may allow deep rooted species to "store" water for use later in the season, preventing shallow rooted competitors from utilizing the water and reducing evaporative losses (Burgess et al. 1998). Another possibility is that the role of hydraulic redistribution may actually facilitate root growth and persistence into other wise dry soils. An increase in soil moisture was shown with the presence of catclaw acacia roots in a deep soil layer (Caldwell et al. 1998).

Study location

The research area lies on the recharge zone of the Carrizo-Wilcox aquifer. This aquifer is comprised of the Carrizo formation and the Wilcox group. The Wilcox Group consists of thick (~600 m) complex sequence of sand silt and clay bodies. The sediments that comprise this part of the aquifer were deposited in fluvial environments that resemble what would currently be described as a braided stream (Fogg 1986). This resulted in a distribution of sands and silty sands inter-bedded with silts and clays. Lateral shifting of stream channels and periodic avulsion of stream courses during deposition lead to lateral heterogeneity (Fogg 1986).

The Wilcox Group is overlain by the Carrizo formation which is laterally continuous and approximately 30 m thick. The depositional environment of the Carrizo formation as described by Hamlin (1983) is an aggrading fluvial system on a coastal plain. It is comprised of two closely interrelated fluvial systems. One system is the "bedload channel" system and the other a "mixed alluvial" system. The massive sand portion of the Carrizo is formed by the bedload channel system and is present in the research area (Hargis 1996). Together, these units form the Carrizo-Wilcox aquifer which is segregated from deeper saline aquifers by underlying shales and carbonates (Fogg and Kreitler 1982).

I have presented a broad overview of the concept of woody encroachment, how it affects ecosystem functioning, and a brief overview of the current state of knowledge relevant to the research question. I have also discussed some of the areas where additional research could provide a more comprehensive understanding to the general issue of woody encroachment as well as a detailed understanding of some below ground processes that are underlying factors of ecosystem function. In the following chapters I will address these issues in two parts. Both of the chapters will follow in manuscript form. The first chapter will broadly address the effect of woody vegetation removal on changes to soil water and the potential to increase deep drainage. The second chapter describes in greater detail the role of plant soil interactions in altering distribution and abundance of soil profile water.

CHAPTER II

INCREASING DEEP DRAINAGE BY WOODY VEGETATION REMOVAL: A STUDY OF SOIL WATER AND ROOTING DEPTH

Introduction

In the Rio Grande Plains of south Texas, woody encroachment has increased the amount of woody vegetation, subsequently altering the ecohydrologic regime of the area. The process of encroachment is initiated by the establishment of a honey mesquite. The mesquite serves as a nucleus to grove clusters that can expand and coalesce to create a closed canopy woodland (Archer et al. 1988). This process is in various stages throughout the Rio Grande Plains. Largely dependent on hill slope location, the region currently is a mosaic of savanna, shrub clusters and closed canopy woodlands. In semi-arid regions like the Rio Grande Plains, the effect of vegetation on the hydrologic cycle can be significant. Currently, it is uncertain to what extent woody vegetation removal can increase deep drainage in mesquite rangelands (Wilcox et al. 2006a). Acquisition of more knowledge is important because shrub and tree encroachment have negative impacts on forage productivity, biodiversity, and socioeconomic potential (Huxman et al. 2005). Additionally, the implications for biogeochemical and hydrologic cycles are poorly understood (Huxman et al. 2005).

Groundwater is an important resource in the Rio Grande Plains. Groundwater accounted for 88% of water used in the Nueces River basin in 2010 (TWDB, http://

www.twdb.state.tx.us/waterplanning/waterusesurvey/estimates/2010). Water not used by vegetation has the potential to recharge the underlying aquifer once it infiltrates the soil and moves beyond the root zone. Water that has moved beyond the root zone is known as deep drainage. Deep drainage occurs when net water inputs exceed the soil storage capacity (Seyfried and Wilcox 2006). On an annual basis, change in soil water storage in semi-arid areas is approximately zero because all the plant available water within the root zone is used (Schwinning et al. 2005; (Seyfried et al. 2005). This is largely due to the fact that potential evapotranspiration exceeds precipitation on an annual basis (Seyfried and Wilcox 2006).

Research has shown that within the mesquite shrubland there are a few soil types conducive to deep drainage if they are in topographic lows or are prone to cracking, but mostly deep drainage is rare (Wilcox et al. 2006a). A review paper and synthesis of the effects of vegetation on groundwater recharge by Kim & Jackson (2012) found that woody plant invasion into grasslands would probably reduce recharge. They concluded that water yield associated with land cover changes varies with climate and soil texture. They found in humid regions and sandy-textured soils the absolute change is likely large but in contrast, changes in deep drainage may be proportionally large in arid or clayey regions. Changes to deep drainage are proportional to and dependent on the relative reduction in vegetation. This effect in scrub lands is largely a function of precipitation, where low rainfall results in only small changes to deep drainage (Bosch and Hewlett 1982).

Brush management has been an ongoing effort in Texas and the southwest. Land managers implement different techniques for various outcomes. Traditionally, the use of rangeland and need for brush management is to improve or maintain livestock production (Scifres 1980). Rangelands provide other environmental services such as furnishing wildlife habitat and increasing recreation potential. More recently there has been an interest in managing woody vegetation for maximum water yield.

Brush management: principles and practices for Texas and Southwest by Charles Scifres (1980) describes a variety of common methods of woody vegetation removal. Woody vegetation is commonly managed using herbicides, fire, mechanical removal, or a combination of methods. Herbicides are applied to the leaves or the soil to kill or retard the growth rate of woody vegetation. Fire is also used as a brush management tool. Controlled burning and prescribed burning are different implementation strategies. Controlled burning is similar to prescribed burning but lacks long term management objectives. Prescribed burning is systematically planned, employs fire control methods and is generally part of a long range management that may include other management strategies. Mechanical removal is categorized by simple top removal or complete plant removal. Examples of top removal are shredding and roller chopping. Examples of complete plant removal are grubbing, chaining, cabling, railing, bulldozing, and root plowing. Desired outcomes, vegetation being treated, and efficacy of treatment are all considerations when selecting a means for managing vegetation.

In Texas, studies on the effect of woody vegetation removal in the water balance in mesquite shrublands have primarily been conducted in the Edwards Plateau, Rolling

Plains, and the Rio Grande Plains regions. Although broadly similar in vegetation and climate, the Edwards Plateau makes a poor analogue for the Rio Grande Plains due to shallow soils and the underlying karst aquifer. In the Rolling Plains region, it was found that essentially no net change to deep drainage, evapotranspiration (ET) or runoff was associated with shrub removal (Carlson et al. 1990). This finding is attributed to the increase in herbaceous vegetation offsetting any decrease in the transpiration of woody vegetation. Heitschmidt and Dowhower (1991) cautioned about extending Carlson's research to other regions because only single stemmed mesquites were studied and the herbaceous response was not measured. A study by Wilcox et al. (2006b) in the Rolling Plains concluded there was little potential to increase deep drainage through removal of woody vegetation in this type of landscape.

Research on woody vegetation removal in the Rio Grande Plains has shown that removal can increase deep drainage by modest but measureable amounts depending on soil characteristics and the degree of reduction in vegetative cover (Moore 2012). A study on loam and sandy loam soils conducted by Weltz and Blackburn (1995) in the Rio Grande Plains concluded that increasing deep drainage or runoff through vegetation manipulation is marginal and limited to years when annual rainfall exceeds potential evapotranspiration. In a paper assessing woody vegetation removal across encroached regions of Texas, Wilcox et al. (2006a) suggested that additional field research is needed to determine the extent to which rangelands in this region have the potential for increased deep drainage following shrub control.

In the Rio Grande Plains and across soil textures, varying densities of woody and herbaceous plants are found together, and one should not be examined independently of the other (Van Auken 2009). To achieve realistic estimates of ecosystem properties and processes, actual rather than assumed root extent must be evaluated on a site specific basis (Stone and Kalisz 1991). In consideration of the variety of soils present in the southwestern portion of the recharge zone of the Carrizo-Wilcox aquifer (see Chapter 1) and the varying and poorly understood effects of roller chopping and top removal on changes to soil water and deep drainage this research aims to address sources of variability such as plant rooting and soil texture in a comprehensive field based experiment.

For this study we established a full factorial field experiment across 3 soil types with 3 woody vegetation removal treatments applied at the beginning of the project to evaluate the effects of vegetation removal on the potential to increase deep drainage. Deep drainage may be thought of as a necessary initial step that may lead to aquifer recharge in the recharge zone, but flow paths beyond 2 m soil depth were not studied here. First, we estimated the average rooting depth to establish the approximate lower limit for water to become potential recharge. Then we measured changes in soil water content within the root zone across soils and removal treatments. Analysis of changes in bulk density, root biomass and soil water content are used to further understand soil and treatment responses.

The removal of woody vegetation is expected to decrease water use and therefore result in an increase in soil water content in deeper soil depths beyond the reach of herbaceous vegetation. Water movement into deeper soil layers is also more likely to occur in sandier soils. Soil disturbance by roller chopping is likely to decrease compaction and allow water to more easily move into the soil profile. For these reasons, we expect to see the greatest response to vegetation removal by roller chopping in sandy soils.

Materials and methods

Study site description

The research site is located in the Northern Rio Grande Plain Major Land Resource Area (28° 56’ 40” N, 100° 3’ 58” W) (NRCS, USDA). The plain is nearly level with smooth gently rolling hills and valleys. Elevation across the region ranges from 60 m in the southeast to 200 m in the northwest. Average annual precipitation ranges from 533 to 939 mm, decreasing from east to west across the region. The average annual temperature ranges from 19.5 to 22 °C.

The potential vegetation of southern Texas and northern Mexico has been classified as *Prosopis Acacia* savanna (Kuchler 1964). Currently, this landscape is dominated by dense woodlands, while some landscapes remain as savanna; others are thought to be still developing into closed-canopy woodlands (Brown and Archer 1990). The variability of soil characteristics interact with rooting patterns and result in a varied abundance and distribution of plant species across these landscapes (Brown and Archer

1990). Some of the common tree and shrub species of the area are *Prosopis glandulosa* Torr. var. *glandulosa*, *Acacia berlandieri* Benth., *Acacia rigidula* Benth., *Acacia schaffneri* (S. Wats) Herm., *Diospyros texana* Scheele, *Aloysia gratissima* (Gill. & Hook.) Troncoso, *Guaiacum angustifolium*, *Acacia gregii*, *Celtis pallida* Torr., *Zizyphus obtusifolia* (T. & G.) Gray var. *obtusifolia*, and *Condalia hookeri*. Grasses common to the area are *Bouteloua curtipendula*, *Heteropogon contortus*, *Pennisetum ciliare*, *Tridens eragrostoides* (Vasey & Scribn.) Nash, *Trichloris pluriflora* Fourn., Digitari *californica* (Benth.) Henr.

Experimental design

This experiment follows a randomized block full- factorial design. Three vegetation removal treatments were applied to three soil textures for a total of nine treatment combinations. These combinations are replicated three times in separate pastures (P1, P2, P3) with different management histories for a total of 27 plots (Figure 2.1). Each plot is approximately 0.10 ha, or 40 m by 25 m. Treatments were randomly assigned. The soils chosen represent the range of textures present in the groundwater recharge zone of the southern Carrizo-Wilcox aquifer (Soil Survey Staff 2012).

Treatment and soil combinations are replicated in three pastures each with varying management histories. The three pastures were selected for consistent management histories, which account for the block effect in the experimental design. Management histories include cattle grazing, mechanical or chemical removal of woody vegetation or combinations of each (Haeglin 2012). Differences in woody plant densities, dominant species, and percent cover of woody and herbaceous vegetation among the three pastures were observed and were broadly consistent with verbal management history descriptions from land managers (personal observation).

Each pasture identified here as Pasture 1-3 (P1-P3; see Chapter III) has a varying management history. P1 has remained relatively untreated for woody encroachment but was grazed under a “high intensity/ low frequency” management system for many decades (Hamilton 2012). P1 has approximately equal percent cover of woody and herbaceous vegetation, and a higher percent bare ground than P3. P2 was subject to extensive herbicide testing in the 1960’s and 1970’s (Hamilton 2012) and the woody vegetation is smaller statured than P3 or P1 with a high percent cover. There has reportedly been no management of woody vegetation on P3 (Haeglin 2012). There was high management turnover on P3, which could result in some vegetation removal to be unreported. Vegetation on P3 is much denser than the other two pastures and has a sparse understory.

These soils were located and selected based on NRCS soil survey maps and include Antosa-Bobillo sand association (sandy), Webb fine sandy loam (sandy loam) and Chacon clay loam (clay loam) (Table 2.1). The finest textured soils studied are the Chacon clay loam. Vegetation on the clay loam sites is generally very dense closed canopy with minimal understory. The intermediate texture soil is Webb series, a sandy loam. Vegetation on the Webb soil is approximately 30% cover of woody vegetation, 30% herbaceous and 40% bare ground. The sandy soils are the Antosa-Bobillo association, where each soil type is not differentiated at the mapped scale. These soils generally exhibit savanna-like vegetation with large-statured mesquite (> 3 m tall), little bare ground, and up to 80% herbaceous cover. The sandy soils also exhibit the greatest variation in vegetation cover, both in percent and species distribution (personal observation).

Table 2.1. Soil descriptions for soils studied. Antosa and Bobillo (sandy), Webb (sandy loam) and Chacon (clay loam) (Soil Survey Staff 2012).

Horizon	Depth (cm)	Texture
Chacon Fine, smectitic, hyperthermic, Torrertic Argiustolls		
A	0-38 cm	clay loam
Bt	38-81	clay
Btk	81-102	clay
Bckyz1	102-132	clay
BCkyz2	132-168	clay
Webb Fine, smectitic, hyperthermic, Aridic Paleustalfs		
A	0-25	very fine sandy loam
Bt1	25-48	sandy clay
Bt2	48-66	sandy clay loam
Btk1	66-97	sandy clay loam
Btk2	97-127	sandy clay loam
Btyz1	127-165	sandy clay loam
Btyz2	165-183	sandy clay loam
Cdz	183-203	soft sandstone bedrock crushes to sandy clay loam
Antosa Loamy, siliceous, active, hyperthermic, Aquic Arenic Paleustalfs		
A1	0-36	sand
A2	36-61	sand
E	61-76	sand
Btg	76-97	sandy clay
Bt1	97-119	sandy clay loam
Bt2	119-183	sandy clay loam
Bobillo Loamy, siliceous, active, hyperthermic, Grossarenic Paleustalfs		
A1	0-76	sand
A2	76-122	sand
E	122-147	sand
Bt1	147-183	sandy clay loam
Bt2	183-203	sandy clay loam

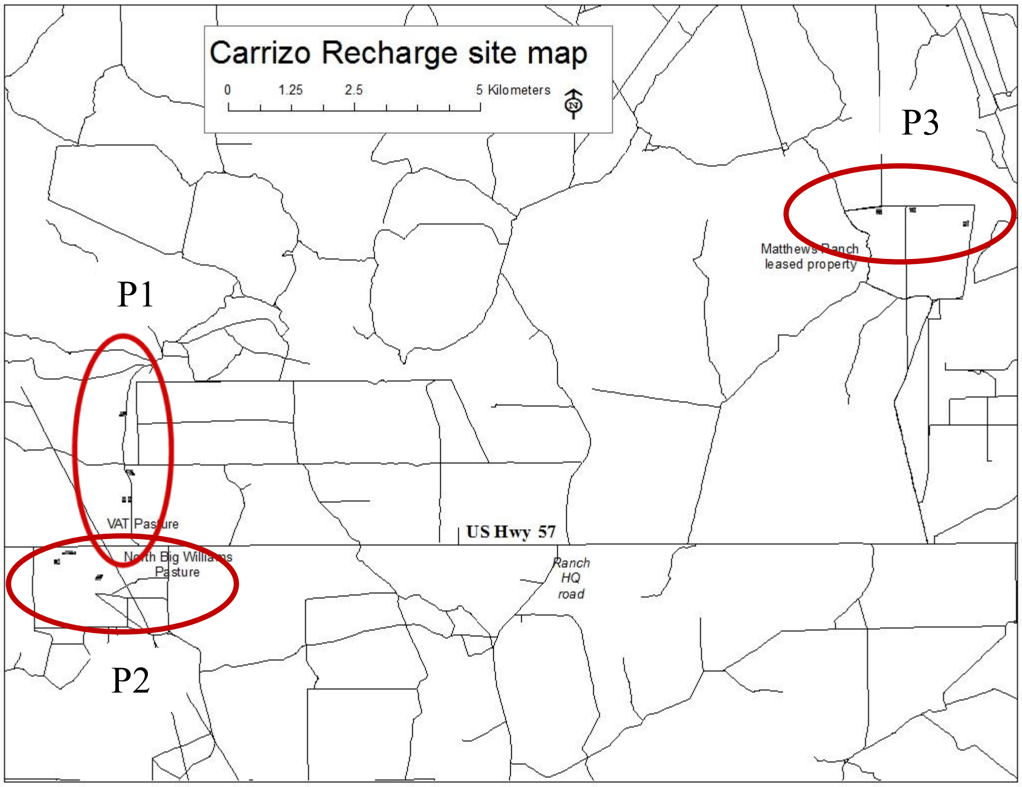

Figure 2.1 Locations of each of the pastures (blocks). Vat Pasture (P1), North Big Williams Pasture (P2) and Mathews Ranch leased property (P3).

Vegetation removal treatments

The vegetation-removal treatments represent a range of effectiveness of woody plant removal and have varying effects on woody and herbaceous vegetation as well as soil disturbance. These vegetation management methods are commonly used and are also recommended by Texas A&M Agrilife Extension (http://texnat.tamu.edu/ about/ brush -busters/cut-stumps/cut-stump-spray-for-hardwood-species/).

Two vegetation removal treatments were chosen, plus a no-removal control. Treatments were applied in fall 2010. The removal treatments were cut-stump and roller chopping. Cut-stump uses a combination of mechanical and herbicide application to remove all woody vegetation and suppress regrowth. The protocol for cut-stump is as follows: 1) remove top growth using a chainsaw; 2) downed woody material is left in place; and 3) apply Remedy ™ herbicide at 15% concentration diluted in diesel oil to the stumps to prevent regrowth. The expected mortality is seven out of ten trees treated (http://texnat.tamu.edu/about/brush-busters).

Roller chopping, also called tandem drum chopping or aeration, is a common top removal treatment for woody vegetation. A large toothed water-filled drum pulled by a tractor (Pasture Aerator, Lawson Mfg. Inc., now RanchWorx, Palm Harbor, FL) was used to cut and crush vegetation at the soil surface. Most of the woody species are expected to produce sprouts from crowns and roots after mechanical top removal, resulting in plants with multiple stems (Schindler and Fulbright 2003). The decrease in overall woody biomass from roller chopping is expected to be 2-3 yrs (Welch et al. 1985).

<u>Data collection</u>

Weather and precipitation measurements. A weather station was erected at P1 in December 2010. The weather station was equipped to record precipitation using a tipping bucket rain gauge (TE525, Texas Instruments, Dallas, TX, USA), photosynthetically active radiation using a LI-190SB Quantum sensor (Li-Cor, Lincoln, NE, USA), temperature and humidity using a HMP45C temperature and relative humidity probe (Campbell Scientific, Logan, Utah, USA), and wind speed and direction using a 03001 R.M. Young Wind Sentry Anemometer and Vane (Campbell Scientific, Logan, Utah, USA). The station was powered by a 12v battery charged by a solar panel. The data were collected hourly, stored on a CR10X data logger (Campbell Scientific, Logan, Utah, USA), and monitored monthly. To account for small-scale variability in rainfall, an additional tipping bucket rain gauge (Hobo UA-003-64 data loggers, Onset, Cape Cod, MA, USA) was installed each remaining pasture (P2 and P3). These rain gauges also collected hourly temperature measurements.

Volumetric water content measurements. Soil water content was measured approximately monthly in all plots. A total of 54 locations (2 within each plot) were sampled to a depth of 180 cm using a model 503DR, neutron moisture meter (NMM) (Campbell Pacific Nuclear, Martinez, CA, USA). Measurements were taken in 20 cm increments to 180 cm deep for the majority of plots. In instances where bedrock was reached before 180 cm (P1, Webb sandy loam at 80 cm, all treatments, & P3, Chacon clay loam, roller chop at 160 cm) measurement stopped at bedrock. Measurements were taken from August 2011 thru February 2012 using 32 s count times, and in March of 2012, count times were reduced to 16 s. A paired t-test was conducted and found that the values obtained for a 32 vs. 16 second count were not significantly different (p=

0.47). The counts obtained from the NMM were converted to volumetric soil water content (VWC, $m^3\ m^{-3}$) using an in-field, access tube and core calibration method.

Calibration of neutron moisture meter. Field calibration was performed for all three soil types. For calibration, an access tube was installed to 210 cm deep and 32 s counts were collected at 20 cm depth intervals. After counts were recorded, soil cores, 5 cm diameter, were collected at depth increments of 10-30, 31-50, 51-70, 71-90, 91-110, etc. to 190 cm using a bucket auger (Giddings Corp., Windsor CO, USA). Soil segments were weighed at field moisture, oven-dried at 105 °C, and re-weighed.

The fitted calibration curve is described by the following equation:

Soil water content ($m^3\ m^{-3}$ soil) = 0.180 x - 0.099, where x is the count divided by the standard count. The root mean squared error of calibration is 0.02.

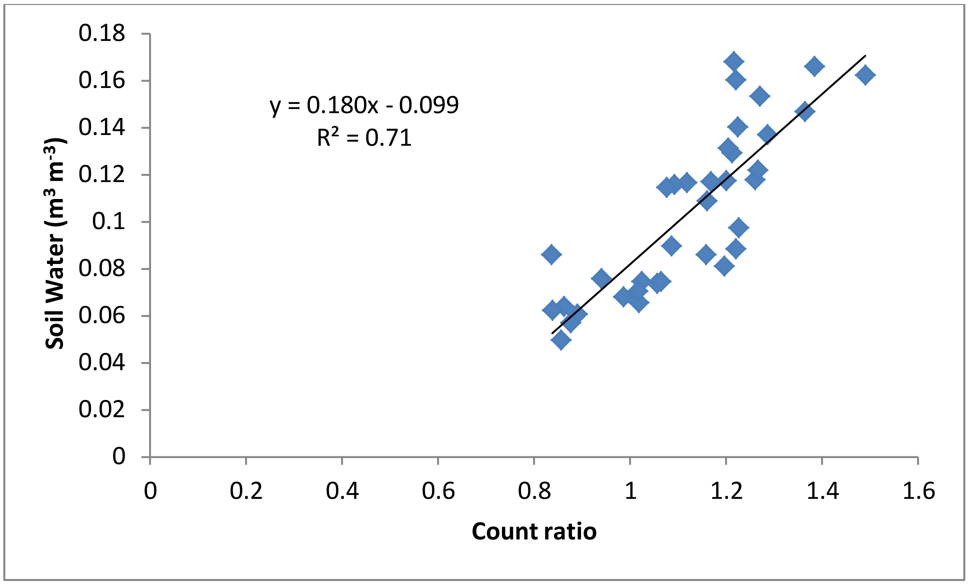

Figure 2.2 Calibration data for NMM.

Typically soils are sampled for calibration of the NMM under dry and wet conditions to capture the range of soil water contents to be measured by the neutron probe. For the calibration used here, only dry soils were sampled. Field conditions limited our ability to sample during wet conditions. The range of soil water contents measured during "dry calibration" as prescribed by the manufacturer is within the range of values measured during our sampling period. During the sampling period, most of the soils remained relatively dry. For this reason the dry calibration alone is assumed to be sufficient to predict soil water content.

Before each round of measurements a new standard count was determined for the moisture meter. To determine the standard count the meter was place on end on top of the lead plate located on top of the carrying case. The case was then placed on top of a

Rubbermaid container to elevate it from the soil so the probe would not be detecting soil moisture during the determination of the new standard count. New standards counts were recorded each month and used to determine the count ratio (count/standard count). Chi square values between 0.75 and 1.25 were considered acceptable at a probability level 95% (IAEA, 2008). When the chi squared value was not within that range, the standard count was redone. A plot of the standard count values over time shows there was no gauge failure throughout the period of measure.

Root biomass and bulk density. During installation of neutron probe access tubes, soils were collected in 20 cm increments using the 5cm diameter bucket auger for a total of 54 cores, 2 from each plot. Known soil volumes were put into soil tins and oven dried soils were weighed in the lab to determine bulk density. Following determination of bulk density, a known weight of the same soil sample was used for determining root biomass on a g cm^{-3} basis. Soil samples were washed using a hydro-pneumatic root elutriator to separate soil from root biomass. Root biomass was oven dried at 40° C, sorted by hand to remove any non-root debris and weighed.

Statistical analysis. Mean soil profile water, changes in soil profile water, root biomass and bulk density were analyzed. Volumetric water content was converted to mm in the 20 cm soil increments and averaged within each replicate. Soil profile water was summed from 0 to 180 cm deep and then averaged for the entire measurement period of 7/28/11 to 9/8/2012 for a total of 13 measurements. Changes in soil profile water were calculated by subtracting the lowest soil water measurement from soil water content at a given measurement time. Lastly, depth intervals of 0-60 cm were defined as shallow soil water, 60-120 cm as a mid soil water and depths of 140-180 cm as a deep soil water. Shallow, mid, and deep soil water layers were used to build a model that analyzed effects of soils, vegetation-removal treatments, and any interactions. Change in soil water was compared over time for shallow, mid and deep soil water as well. Shallow soils were chosen to depth of 60 cm to assess effects of evaporation following Weltz & Blackburn (1995). Mid soils were analyzed to evaluate the influence by roots of woody vegetation (Weltz and Blackburn 1995). Deep soil water was defined as at or near the bottom of the root zone. Analysis of variance was used to test for differences in means (JMP, Version 10 SAS Institute Inc., Cary, NC, 1989-2007). Where appropriate, means were separated using Tukey's mean separation ($P < 0.05$).

Results

Root biomass

Shallow root biomass had a significant soil x vegetation removal treatment interaction at 0-20 cm soil depth (Table 2.2). There was not a significant response for 20-40 or 40-60 cm soil depth. Root biomass data were log-transformed before ANOVA analysis to account for non-normal distribution of residuals. Root biomass at 0-20 cm had a significant soil x vegetation removal treatment interaction ($p= 0.002$, Table 2.3).

Root biomass was greatest for sandy soil, cut stump plots and clay loam roller chop. Root biomass was the least for sandy soil, roller chop. Deep root biomass (120-180 cm) was not significantly different for soil, vegetation removal treatment or soil x vegetation removal treatment interaction, $p = 0.28$.

Table 2.2 ANOVA results for root biomass from 0-20 cm soil depth.

Source	DF	Mean Square	F Ratio	Prob > F
Soil	2	0.205	0.310	0.737
Treatment	2	0.047	0.071	0.932
Soil x Treatment	4	4.493	6.812*	0.002

Table 2.3 Root biomass results (g m^{-2}) of post hoc analysis for significant soil x treatment interaction 0-20 cm soil depth. LS Mean values here are reported as (e^{LSmean}). Soil x treatment combinations with different Tukey's letters are significantly different.

Soil	Treatment	Tukey's	Least Sq Mean	Std Error
Sandy	Cut stump	A	40.0	0.469
Loam	Roller chop	A,B	35.5	0.574
Clay loam	Roller chop	A	33.6	0.406
Sandy	Control	A,B	27.3	0.469
Loam	Control	A,B	15.9	0.469
Clay loam	Cut stump	A,B	10.62	0.469
Loam	Cut stump	A,B	9.18	0.469
Clay loam	Control	A,B	5.90	0.469
Sandy	Roller chop	B	2.39	0.574

Rooting biomass decreases logarithmically with depth across all soil types (Figure 2.3, Figure 2.4, Figure 2.5). Root biomass data were ln-transformed and linear regressions were fit to the data to estimate maximum rooting depth for each soil type. For all regressions, both slope and intercept were significant ($p < 0.001$). Y- intercepts for sandy, loam, and clay loam soils are 195, 171, and 177 cm, respectively. Although the slope and y-intercept for each regression were significant, root biomass distributions are poorly explained by regression analysis. The root distributions across soil depth and soil texture are not explained well by the regression as shown by the r^2 values. The r^2 values for each soil type are 0.30, 0.23, and 0.25 for ABC, WEB, and CKB respectively.

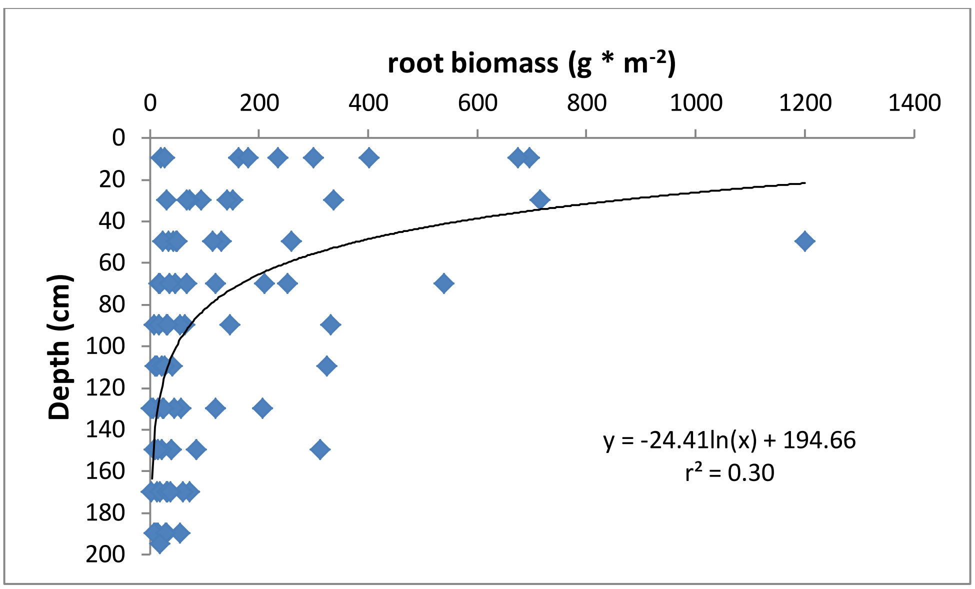

Figure 2.3 Root biomass vs. depth for sandy (ABC) soils. Y-intercept of regression predicts rooting depth.

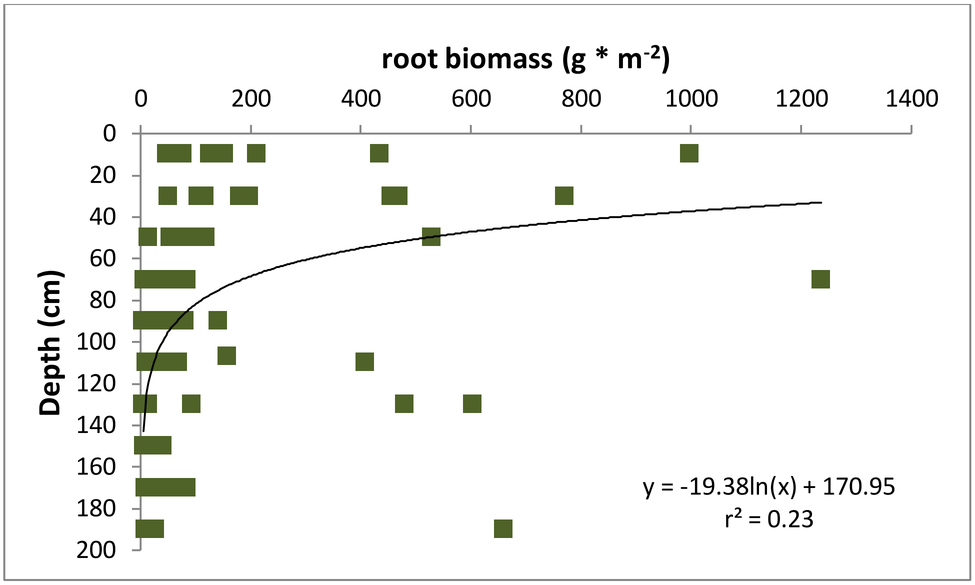

Figure 2.4 Root biomass vs. depth for loam (WEB) soils. Y-intercept of regression predicts rooting depth.

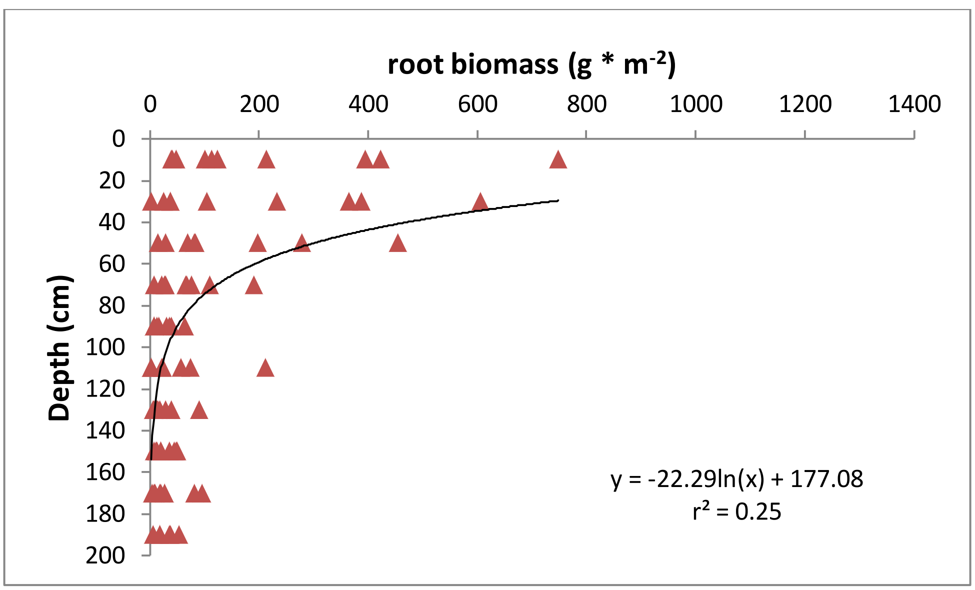

Figure 2.5 Root biomass vs. depth and clay loam (CKB) soil type. Y- intercept of regression predicts rooting depth.

The depths where soils contained 50% and 90% of total root biomass were similar for each soil type (Table 2.4). The root biomass in the sandy soil reached 50% at a depth of 50 cm, and 90% at 140cm. The root biomass in the sandy loam soil reached 50% at a depth of 80 cm, and 90% at 160cm. Root biomass in the clay loam soil reached 50% by 40 cm depth and 90% by 160 cm depth.

Table 2.4 Root biomass by depth and soil type. B_{90} and B_{50} are the depths where 90% and 50% of root biomass occurs, respectively.

	Biomass (g * m^{-2})	Biomass (g * m^{-2})	Biomass (g * m^{-2})
Soil Depth	**ABC**	**WEB**	**CKB**
0-20	300	264	246
21-40	202	294	225
41-60	212	127	105
61-80	146	289	74
81-100	79	129	27
101-120	54	141	57
121-140	56	349	28
141-160	50	34	30
161-180	26	58	36
181-200	27	112	26
Total Biomass	1152	1796	855
$B_{90\ (depth,\ biomass)}$	(140,1037)	(160,1616)	(160,769)
$B_{50\ (depth,\ biomass)}$	(50,576)	(80,898)	(40,427)

Bulk density

Bulk densities of the shallow soil layers were compared at three soil depths of 0-20, 20-40, and 40-60 cm. Zero to 20 cm had a significant soil, treatment and soil x treatment interaction (p=0.0005, Table 2.5). Depths of 20-40 and 40-60 cm did not have significantly different soil bulk densities (p=0.60 and 0.15, respectively). Tukey's post hoc was used to determine mean differences for depth of 0-20 cm (Table 2.6). Sandy soil roller chop and clay soil control had significantly lower bulk densities than sandy control. Bulk densities of the deep soil layers were significantly different by soil type (p = 0.035). For deep soils mean values were 1.59, 1.55, and 1.34 g cm^{-3} for sandy, sandy loam, and clay soils respectively, where the bulk density of the clay soil was significantly lower than the other two soils.

Table 2.5 ANOVA table results for 0-20 cm soil depth bulk density. F-ratios with asterisk are significant (p<0.05).

Source	DF	Mean Square	F Ratio	Prob > F
Soil	2	0.141	8.753*	0.002
Treatment	2	0.072	4.458*	0.027
Soil x Treatment	4	0.105	6.508*	0.002

Table 2. 6 Soil bulk densities (g m^{-2}) for post hoc analysis at 0-20 cm soil depth. Tukey's with different letters within the same effect are significantly different ($p<0.05$).

Effect	**Level**	**Tukey's**	**Mean**	**Std Error**
Soil	sand	A	0.954	0.042
Soil	loam	A	0.928	0.042
Soil	clay loam	B	0.726	0.042
Treatment	Cut Stump	A	0.939	0.042
Treatment	Control	A,B	0.900	0.042
Treatment	Roller Chop	B	0.769	0.042
Soil x Treatment	sand, Control	A	1.105	0.073
Soil x Treatment	loam, Control	A,B	1.078	0.073
Soil x Treatment	sand, Cut Stump	A,B	1.025	0.073
Soil x Treatment	loam, Cut Stump	A,B	0.924	0.073
Soil x Treatment	clay loam, Cut Stump	A,B,C	0.869	0.073
Soil x Treatment	clay loam, Roller Chop	A,B,C	0.790	0.073
Soil x Treatment	clay loam, Roller Chop	A,B,C	0.783	0.073
Soil x Treatment	sand, Roller Chop	B,C	0.733	0.073
Soil x Treatment	clay loam, Control	C	0.518	0.073

<u>Mean soil profile water</u>

Soil water from 0 to 180 cm depth was summed for each profile and averaged by replicate. Soil profile water means of the entire sampling period were then compared across soil and vegetation removal treatments (Table 2.7). Within all roller chop treatments, profile soil water ranked sand > loam > clay loam soils, where sand is significantly greater than clay loam (Table 2.8). Within all cut stump treatments, profile soil water ranked clay loam > loam > sandy soils, where clay loam is significantly greater than sandy soils. Within control plots profile soil water ranked clay loam > sand

= loam. Roller chop plots show a slight increase in total soil water content from Oct. 2011 to Feb. 2012 (Figure 2.6). Cut stump and controls do not show an increase during this time (Figure 2.7, Figure 2.8). Within the sand and loam soils, there was no difference between control plots and treated plots in total soil water content (Table 2.8). Sandy control and treated plots were significantly wetter than only the clay loam roller chop plots.

There was no significant difference in soil profile water within or between sandy and loam soils regardless of treatment. The strongest response to treatment was for clay loam roller chop plot. The clay loam roller chop plot was significantly drier than the clay loam control and clay loam cut stump. The clay loam control was significantly wetter than all the sandy and loam soils. The clay loam roller chop was significantly drier than all the sandy soils (Table 2.8).

Table 2.7 ANOVA results of mean total soil water content (mm) to a depth of 180 cm.

Effect	DF	Mean Square	F Ratio	Prob > F
Soil	2	27590	8.201*	0.0003
Treatment	2	66270	19.70*	<.0001
Soil x Treatment	4	52280	15.54*	<.0001

Table 2.8 Tukey's post hoc of total soil water profile (mm). Means with differing letters are significantly different.

Soil	Treatment	Mean	Std Error
Clay loam	Control	258a	5.327
Clay loam	Cut Stump	248a,b	5.327
Loam	Cut Stump	209b,c	5.327
Sand	Roller Chop	203c	5.327
Sand	Cut Stump	203c	5.327
Sand	Control	189c	5.327
Loam	Control	175c,d	5.327
Loam	Roller Chop	173c,d	5.327
Clay loam	Roller Chop	144d	5.327

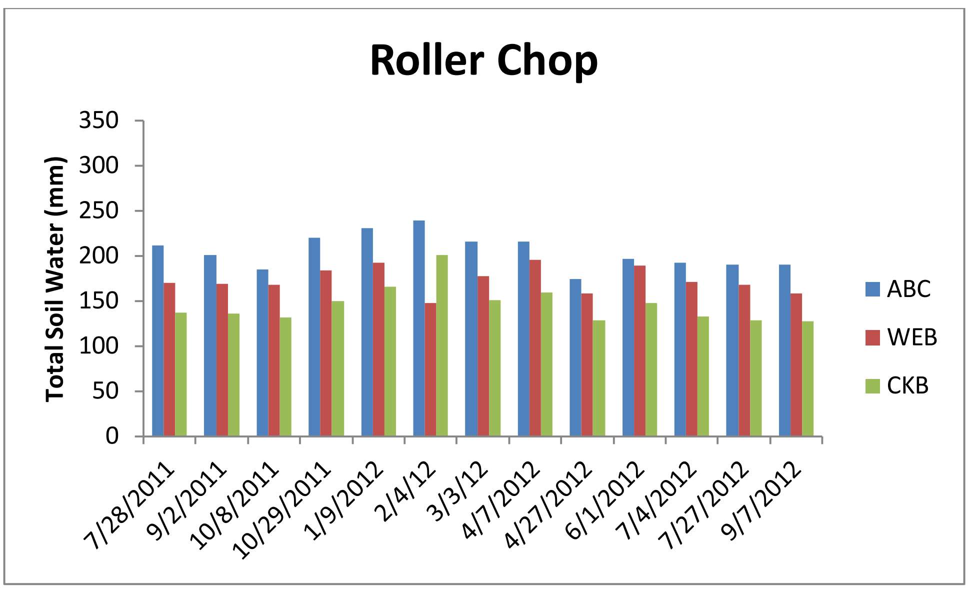

Figure 2.6 Mean soil water content by month and treatment to 180 cm depth.

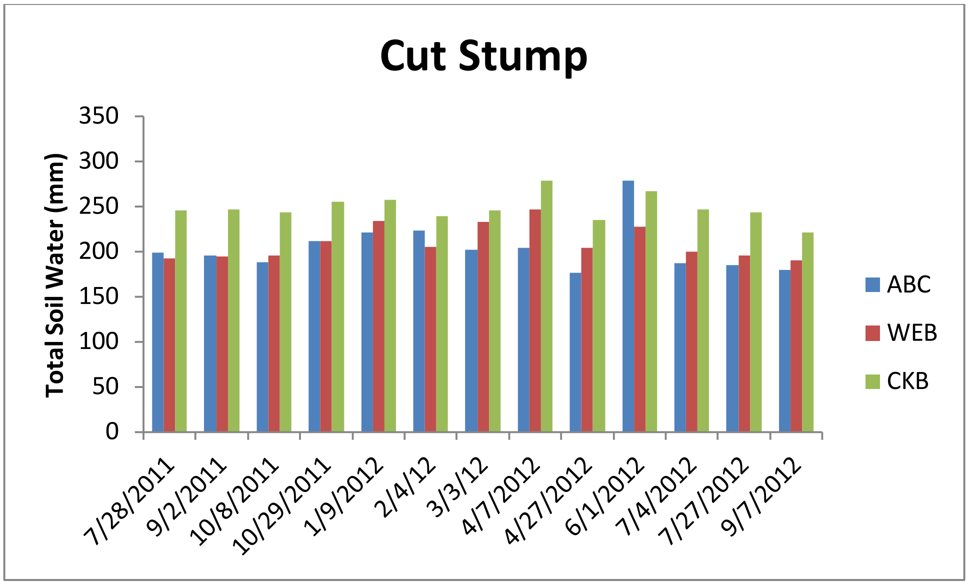

Figure 2.7 Mean soil water content by month and treatment to 180 cm depth.

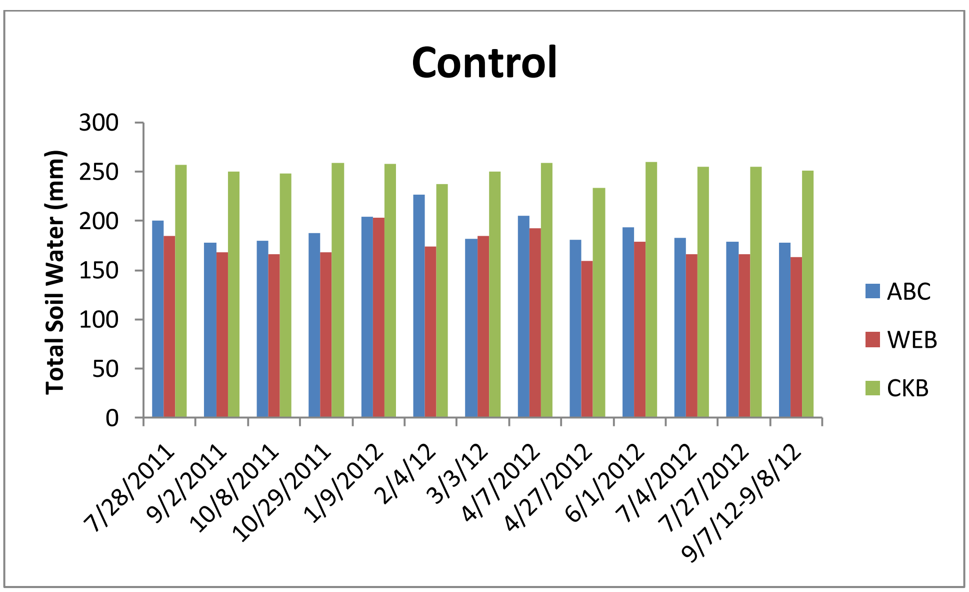

Figure 2.8 Mean soil water content by month and treatment for 180 cm depth.

Change in soil profile water

Each soil type is expected to have different total soil water content due to soil texture regardless of treatment. To account for these differences, the minimum measured soil water content was considered plant unavailable. The stored soil water was subtracted from each monthly measurement to calculate change in soil water content. Change in soil water content was calculated in 20 cm depth increments for each soil and vegetation removal treatment.

Changes in soil water content, for the three depth increments, were analyzed by month. Out of the 13 months analyzed, three months were significant in the shallow soils (0-60 cm), eight were significant in the mid depth soils (60-120) and three months were significant in the deep soils (120-180) (Table 2.9). Change in soil water was significant for all depths on April 2012.

Shallow soils had significant soil effect for three months. There was no vegetation removal treatment effect for any month in the shallow soils. There was no soil x treatment interaction for any month in the shallow soils. The change in soil water content in shallow soils was significantly lower in sand than clay loam soils for three months. Sand was significantly lower than loam soils for two of three significant months (Table 2.9).

Mid depth soils had significant effect for eight months. Five of the eight months had significant soil x vegetation removal treatment interaction. Four of those five months the greatest difference was in the sandy roller chop treatment. Where there was a significant soil x vegetation removal treatment interaction, the soil was the significant factor. Two months had significant soil effect and the change in soil water content was significantly greater in sand than loam and clay loam soils. One month had significant soil, vegetation removal treatment and soil x vegetation treatment interaction effects (Table 2.9).

Deep soils had only one month with a significant difference in change in soil water content for soil x vegetation removal interaction effect. Sandy roller chop had the largest change, but was only significantly larger than clay loam control. The total increase in soil water content for April of 2012 was 39 mm for 120-180 cm.

Table 2.9 ANOVA results for changes in soil moisture by soil depth by month. F-ratios with asterisk are significant ($p< 0.05$).

Depth (cm)	Date	Source	DF	Mean Square	F Ratio	Prob > F
0-60	10/8/2011	Soil	2	41.781	4.033*	0.022
		Treatment	2	8.595	0.8296	0.440
		Soil*Treatment	4	5.784	0.5583	0.694
0-60	4/7/2012	Soil	2	308.931	7.369*	0.001
		Treatment	2	23.333	0.5566	0.576
		Soil*Treatment	4	35.594	0.849	0.499
0-60	6/1/2012	Soil	2	179.720	6.153*	0.003
		Treatment	2	26.141	0.895	0.413
		Soil*Treatment	4	21.265	0.728	0.576
60-120	7/28/2011	Soil	2	281.711	9.006*	0.000
		Treatment	2	45.784	1.4636	0.235
		Soil*Treatment	4	46.225	1.4777	0.212
60-120	9/2/2011	Soil	2	66.188	3.318*	0.039
		Treatment	2	25.135	1.2598	0.287
		Soil*Treatment	4	50.260	2.519*	0.044
60-120	1/9/2012	Soil	2	333.201	8.008*	0.001
		Treatment	2	148.701	3.574*	0.031

Table 2.9 Continued

Depth (cm)	Date	Source	DF	Mean Sq	F Ratio	Prob >F
		Soil*Treatment	4	116.753	2.806*	0.028
60-120	2/4/2012	Soil	2	241.268	5.890*	0.004
		Treatment	2	13.742	0.3354	0.716
		Soil*Treatment	4	46.222	1.1282	0.348
60-120	3/3/2012	Soil	2	176.702	5.488*	0.005
		Treatment	2	265.177	8.236*	0.000
		Soil*Treatment	4	190.082	5.904*	0.000
60-120	4/7/2012	Soil	2	69.991	2.2399	0.110
		Treatment	2	63.352	2.0274	0.135
		Soil*Treatment	4	133.176	4.262*	0.003
60-120	7/4/2012	Soil	2	3.591	0.208	0.813
		Treatment	2	12.713	0.7363	0.481
		Soil*Treatment	4	64.312	3.725*	0.006
60-120	9/7/2012	Soil	2	3.768	0.207	0.813
		Treatment	2	34.158	1.8765	0.157
		Soil*Treatment	4	59.361	3.261*	0.014
120-180	9/2/2011	Soil	2	119.676	1.2156	0.300
		Treatment	2	4.741	0.0482	0.953
		Soil*Treatment	4	341.508	3.469*	0.010
120-180	4/7/2012	Soil	2	245.785	2.4173	0.093
		Treatment	2	46.025	0.4527	0.637
		Soil*Treatment	4	313.991	3.088*	0.018
120-180	9/7/2012	Soil	2	133.357	1.5571	0.214
		Treatment	2	32.898	0.3841	0.682
		Soil*Treatment	4	257.299	3.004*	0.021

Table 2.10 Tukey's results of post hoc analysis of differences in change in soil water content by month for shallow, mid and deep soils. Means (mm) with differing letters are significantly different. Dashes indicate non-significant differences (see Appendix C for complete soil moisture results)

		ABC	WEB	CKB	RC	CS	Cntrl	ABC	WEB	CKB	Tx
0-60	10/7/11	0.87b	3.13a	2.91a,b	-	-	-	-	-	-	-
0-60	4/7/12	4.91b	11.37a	9.88a	-	-	-	-	-	-	-
0-60	6/1/12	2.76b	7.74a	6.42a	-	-	-	-	-	-	-
60-120	7/28/11	5.54a	1.75b	1.38b	-	-	-	-	-	-	-
60-120	9/2/11	-	-	-	-	-	-	4.18a,b	1.59b	4.20a,b	Cntrl
		-	-	-	-	-	-	3.29a,b	3.85a,b	2.20b	CS
		-	-	-	-	-	-	7.23a	3.77a,b	2.23b	RC
60-120	10/29/11	-	-	-	-	-	-	4.46a,b	1.66a,b	32.56a	Cntrl
		-	-	-	-	-	-	5.17a,b	4.21a,b	2.24b	CS
		-	-	-	-	-	-	11.95a,b	3.81a,b	3.84a,b	RC
60-120	1/9/12	7.22a	2.91b	2.84b	6.21a	3.88a,b	2.88b	4.46b	1.02b	3.16b	Cntrl
		-	-	-	-	-	-	5.17b	4.75b	1.71b	CS
		-	-	-	-	-	-	12.02a	2.94b	3.66b	RC
60-120	2/4/12	8.23a	4.21b	3.34b	-	-	-	-	-	-	-
60-120	3/3/12	5.73a	3.23a,b	2.20b	5.71a	4.24a	1.21b	1.08b	0.65b	1.89b	Cntrl
		-	-	-	-	-	-	4.57b	6.42a,b	1.73b	CS
		-	-	-	-	-	-	11.52a	2.62b	2.97b	RC
60-120	4/7/12	-	-	-	-	-	-	6.11a,b	2.84b	4.91a,b	Cntrl
		-	-	-	-	-	-	4.53a,b	9.11a	4.75a,b	CS
		-	-	-	-	-	-	10.37a	5.54a,b	4.52a,b	RC
60-120	7/4/12	-	-	-	-	-	-	5.16a	1.79a	5.02a	Cntrl
			-	-	-	-	-	1.86a	5.14a	2.83a	CS

Table 2.10 Continued

		ABC	WEB	CKB	RC	CS	Cntrl	ABC	WEB	CKB	Tx
		-	-	-	-	-	-	5.24a	4.55a	2.86a	RC
60-120	9/7/12	-	-	-	-	-	-	4.50a	1.35a	4.84a	Cntrl
		-	-	-	-	-	-	1.17a	4.05a	1.59a	CS
		-	-	-	-	-	-	4.81a	3.82a	2.58a	RC
120-180	9/2/11	-	-	-	-	-	-	3.30a	11.67a	6.57a	Cntrl
		-	-	-	-	-	-	9.52a	5.95a	7.89a	CS
		-	-	-	-	-	-	13.46a	6.33a	3.03a	RC
120-180	4/7/12	-	-	-	-	-	-	4.61a,b	12.37a,b	2.62b	Cntrl
		-	-	-	-	-	-	9.29a,b	6.61a,b	9.44a,b	CS
		-	-	-	-	-	-	13.29a	7.03a,b	3.16a,b	RC
120-180	9/7/12	-	-	-	-	-	-	3.24a	11.90a	6.95a	Cntrl
		-	-	-	-	-	-	8.62a	6.29a	3.88a	CS
		-	-	-	-	-	-	12.44a	6.16a	5.01a	RC

Discussion

Rooting depth

Rooting depth is an important determination to be made for this ecosystem. Water that moves beyond the roots is available to become potential recharge. Water that is within the root zone has the potential to be lost by transpiration or evaporation. Changes in water content within the root zone may be affected by woody vegetation removal treatment or soil type, but do not necessarily lead to deep drainage.

Vegetation removal treatments were completed in late 2010. Root biomass sampling occurred from 5-9 months following treatments. Statistical analysis showed that there was not a significant treatment effect on root densities in the deep soil layer. For this reason, all the root biomass data for all vegetation removal treatments (and control) was pooled in the determination of rooting depth. The rooting depth presented is for the entire ecosystem, including shrubs and herbaceous component.

ANOVA analysis showed no significant difference in root biomass below depths of 120 cm. This suggests that roots are very low density and differences are difficult to detect or there was not enough time following treatment to see a reduction in rooting depth. Slow growing species in low nutrient conditions are known to increase tissue density which reduces the decay rate (Ryser 1996). Root decay rates may have been exceptionally low during this time due to very low water content in the soil. If changes do result from treatments at this depth, it may be difficult to detect due to slow decomposition rates in the deep soil layers. Decomposition rates at this depth may be very low and roots may persist for a long period of time following treatments and have a residual effect on soil water content by acting as preferential flow paths.

Regression analysis estimated average rooting depth to range from 170-194 cm, but the fit of the lines, although significant were very poor. Depth estimates for 90% root biomass in sand, loam and clay loam soils were 140, 140, and 160 cm respectively. Moore et al. (2010) estimated roots in the same system to have 95% root biomass at 150 cm. Heitschmidt et al. (1988) estimated 90% of root biomass to exist above 133 cm. Watts (1993) estimated 92% of root biomass to occur above 120 cm. The determination made by this analysis is consistent with other research and estimates root biomass, regardless of soil type in this system to reach 90 % between approximately 120 and 180 cm.

Statistical analysis of root biomass and bulk density showed significant differences in 0-20cm soil depth. The lowest root density was in the sandy roller chop and the highest root densities were in the sandy cut stump and the clay loam roller chopped. The low values for sandy roller chop could be attributed to large soil disturbance and high decay rates in exposed soil. This could also be due to low initial root biomass in 0-20 cm in the sandy soil. This does not explain why loam and clay loam soils did not have lower root biomass in the roller chop plots. This is somewhat contradictory to sandy cut stump having one of the highest root densities.

It would be reasonable to expect control plots to be the highest, followed by cut stump and then roller chop, but this is not the pattern. A possible explanation for high values in clay loam roller chop plots is organic matter being incorporated into the soil during roller chopping and misidentification of grasses as roots. Again, this does not explain why sandy and clay loam roller chop values were opposite. Low root biomass values in clay loam control and cut stump plots could be due to loss of fine roots during root elutriation. Also, because roots were measured on a mass basis only, a single large root could overshadow the existence of many fine roots in a soil type or treatment. The root biomass data gives a reasonable estimate of rooting depth; however, the root biomass data has not been useful in understanding smaller scale and shorter term responses to woody vegetation removal.

The bulk density of the control plots is consistent with expected bulk density values, where clay loam has the lowest bulk density and sandy and loam soils have higher bulk densities. This pattern hold true for the cut stump plots where there was no soil disturbance from vegetation removal. Roller chop treatment causes significant soil disturbance in the top 20 cm. Following roller chopping, clay loam had the highest bulk density, followed by loam and sandy soils. A study on bulk density in a sandy loam found that after plowing, there was decrease in compaction, but soils returned to equilibrium soil density within a single growing season (Carter 1990). This helps to explain the lack of change for the loam soil bulk densities. Tillage practices can alter pore size distribution, soil structure and stable macropores (Buckley et al. 2010). This could explain the longer term changes to clay loam and sandy soils. Compaction by roller chopping in the clay loam soil could also account for the increase in bulk density.

The decrease in bulk density for sandy soils was visually apparent following roller chopping.

Changes to bulk density in the shallow soils due to treatment alter the time needed for water to move into deep soil, away from the effects of evaporation, and beyond the majority of roots. An increase in bulk density should decrease infiltration rates and quantities. A decrease in bulk density would have the opposite effect. This is what we see in the sandy roller chop plots where there is a larger change in soil water at mid and deep soil depths compared to other soils and vegetation removal treatments. Changes in bulk density as a result of roller chopping could also increase the time and decrease the amount of water moving through the shallow soils into deeper soil depths in the clay loam soils. This is consistent with the reduced overall soil water content for the clay loam roller chop plots. The disruption of the root network and reduction in hydraulic descent is suggested as another major mechanism for the reduction in soil profile water. Changes as a result of alteration to the process of hydraulic lift will be discussed in detail in the following chapter.

Soil water

Differences between total soil water content is not necessarily a direct response to treatment, but differences in soil water storage based on soil texture. When we compared total soil water content across soil types, it is not surprising that clay loam controls are wetter than sandy and loam soil controls. Clay loam soils have higher field capacities than sandy and loam soils and have more negative water potentials at the same water content. Due to physiological constraints, plants will only be able to extract water from a soil to a minimum value. If the lowest values measured in each soil are a result of loss of water by vegetation, clay loam is still expected to have higher water content. In the shallow soils (0-60 cm), there was only a significant soil effect. The changes in soil water content were the lowest in the sandy and greatest in clay loam soils. This is consistent with the low holding capacity and the high hydraulic conductivity of sands and the inverse for clays. There was no vegetation removal treatment effect. This is not surprising, because soil texture is such a strong factor in soil water, so if a treatment effect were significant, it would result in a significant soil x vegetation removal interaction effect. There was no significant soil x treatment interaction in the shallow soils. This supports the findings of Carlson et al. (1990) that suggests that the differences in treatment response are muted by changes in ET. Weltz and Blackburn (1995) found that there is no difference in ET for shrub clusters or grass interspaces. The increased plant available water due to the removal of woody vegetation reduces water competition for shallow rooted herbaceous vegetation. Essentially, the herbaceous component utilizes the water that is not used by the shrubs resulting in a significant soil effect in water content in the shallow soil.

There were significant soil x treatment interactions in the mid depth (60-120 cm) soils. Soil was always the dominant factor, and in only one of seven months was vegetation removal treatment also significant. This suggests that treatment does not have a strong control on soil water content or changes in soil water content. Sandy roller chop had one of the greatest changes in soil water content at mid depths. Following roller chopping the shallow soils had a very low holding capacity due to reduction in bulk density. The total amount of water in the soil profile did not change in response to roller chopping in the sandy soil, only the location of the water. Water moved quickly thru the shallow soils and as a result, there were greater changes in the mid depth soils.

There was the least amount of change in the mid depth in the clay loam roller chop soils. There was an increase in bulk density in clay loam soils following roller chopping, but this poorly explains the changes in soil profile water. It should be expected, that increases in bulk density in shallow soils would decrease the amount of water in the mid depths because water is being held in the shallow soils, but the total soil profile should not change. However, there was an overall reduction in soil profile water, not just a change in distribution across the profile. It could be reasoned that water was lost to runoff or evaporation in the shallow soils before it could percolate to deeper soils. Brock et al. (1982) found in a clay loam, infiltration rates were higher and runoff rates were lower following two vegetation removal techniques similar to roller chopping and cut stump. Tan et al. (2002)found a tillage effect on runoff during the non- cropping season, and no tillage effect on runoff during the cropping season, suggesting, vegetation is having some effect on the movement of water over as well as into soils.

The purpose of this research is to determine the potential to increase groundwater recharge by woody vegetation removal. Water moving beyond the root zone, known as deep drainage, is available to become potential recharge. Changes in soil water in the mid depth (60-120cm), although significant, do not directly influence potential recharge.

Analysis of rooting depth shows that the majority of roots occur above 120-180 cm soils depth. Changes in soil water content in the deep soil horizons are the most indicative of changes to potential recharge.

April 2012 was the only month that had a significant interaction in the deep soil. There was no significant main effect. The only significant differences were between sandy roller chop and clay loam control, where the sandy soil had the greatest change in soil water content, and clay loam soil had the smallest change. Sandy roller chop had 30 mm more water at a depth of 120-180 cm than clay loam control. This was the greatest difference between any of the soil x vegetation removal treatment interactions. Across 60 cm of soil this is a difference of 5%. An increase of 5% soil water should not be enough water to create deep drainage considering all of the soils are clayey at that depth and had volumetric water contents well below field capacity.

Based on the patterns produced by graphs of VWC over time with depth for ABC soils, it appears that the clay horizon begins on average around 100 cm, and continues to 200 cm. If the average soil water content for that horizon is 0.16 cm^3 cm^{-3}, and the value needed to exceed field capacity for the sandy clay is 0.31 cm^3 cm^{-3}, then at any given time, there is a deficit of 0.15 cm^3 cm^{-3} for that soil horizon needed to result in gravitational water moving through. This amounts to 387 mm of precipitation, slightly over half of the annual average of rain for the area. Field capacity of soils are roughly 0.15 cm^3 cm^{-3} and 0.30 cm^3 cm^{-3} for sand and clay loam soils respectively (Walker and Skogerboe 1987). Soil water content did exceed those needed for gravitational flow in the sandy horizon, but did not even approach those needed in the clay horizons in deeper soils. At no point during the study period was soil water content near values for the clay loam soils to exceed field capacity. So, in spite of the measureable differences in response to treatments across soil type, the difference is not likely to be great enough create deep drainage.

Another noteworthy response to woody vegetation removal is the decrease in total soil water content in the clay loam roller chopped plots. Interestingly, the clay loam cut stump was not significantly lower than the control, and had one of the highest total soil water contents. Based on soil water holding capacity, we would expect the clays to have a higher overall water content, but this does not explain the low water content for the clay loam roller chop plots. One explanation might be the increase in soil bulk density increases runoff. Runoff was not measured, but field observations did not indicate any increase in runoff. Research has found that on a similar clay loam under no treatment runoff is <1% of the water budget, but does not account for effects of roller chopping (Wilcox et al. 2006a). On a clay loam, Tan et al (2002) found that a no till treatment (similar to cut stump in terms of soil effects) increased runoff whereas moldboard plowing (comparable to roller chopping) decreased runoff. These results are opposite of what is necessary to explain the differences in soil water content for clay loam control, cut stump and roller chop treatments. Another explanation for the decrease in soil water content is a decrease in macropores and soil structure that inhibits soil water from moving through the soil and is thus lost to evaporation in the shallow soil (Buckley et al. 2010). Alternatively, if hydraulic redistribution is a factor influencing soil water movement into the soil, the destruction of a shallow root network in the clay soils could obstruct a pathway for water into the soils. The mechanisms and underlying factors that are controlling the changes in soil water content associated with treatment and soil type are discussed further in the following chapter.

Conclusions

Rooting depth within our sampling region in the northern Rio Grande Plains region is approximately 150 cm across a range of soil textures. Average amounts of annual precipitation in combination with the effects of soil characteristics and vegetation on soil water movement limit the likelihood of deep drainage occurring under any vegetative cover. For deep drainage to occur, soils must be saturated beyond the depth of roots. Because rainfall is evenly distributed throughout the year, water that enters the soil is quickly lost by evaporation or transpiration.

The combination of roller chopping and sandy soils results in the greatest change in soil water content at 60-120 cm soil depth. Under average rainfall of ~ 600 mm annually, roller chopping on sandy soils is unlikely to result in water moving beyond the root zone and becoming available for recharge due to soil physical limitations in the clay horizon and plant water uptake. Destruction of roots in shallow clays soils could limit connectivity of shallow and deeper soils via a root network, thus eliminating a major water pathway of water into clayey soils. The effect of roller chopping on clay loam soil has the potential to decrease the amount of water moving beyond the root zone and becoming available for recharge. Due to high evaporative and transpirational demand by woody and herbaceous plants in combination with soil physical limitations and limited rainfall, the likelihood to increase deep drainage by woody vegetation removal in the recharge zone of the Carrizo-Wilcox aquifer is very low.

CHAPTER III

THE PATHWAY OF WATER IN AND OUT OF SOILS: THE ROLE OF ROOTS IN A SEMI-ARID WOODY ENCROACHED ECOSYSTEM

Introduction

Within a soil profile, soil texture and structure determine the amount of plant available water. Water can also move into soils by way of gravity through the soil matrix, by preferential flow or by changes in soil matric potential (Bouma 1981). Hydraulic redistribution, another way water moves within the soil, is passive water movement along a gradient from higher to lower water potential through roots acting as conduits (Burgess et al. 1998, Caldwell et al. 1998). Vegetation type can also influence how water moves into and within the soil. Coarse-textured soils generally permit faster rates of infiltration and deeper percolation. Coarse-textured soils should favor deep rooted woody plants exploiting water at greater depths. Fine-textured soils limit water penetration to lower horizons and retain water in the upper soil layers. Fine-textured soils should favor shallow rooted grasses (Brown and Archer 1990). The root distribution within a soil profile has a large influence on soil water content at various depths. In shallow soils, the amount of water utilized by the plant is often high because root abundance is typically greater (Midwood et al. 1998). Water use by plants at greater soil depths is usually lower where root abundance is lower (Le Roux et al. 1995). Along streamsides, or where roots have access to groundwater, plants can access significant abundances of water in spite of low root abundance (Canadell et al. 1996, Hultine et al. 2004). In some instances grasses acquire water only from the upper soil layer, and woody vegetation has exclusive access to the deeper soil layers thus partitioning soil water use (Le Roux et al. 1995). Another pathway for water is woody plants moving water from shallow to deep soil horizons. Water moved into the soil by woody plants may in some cases reach depths that would not be reached by Darcian flow alone (Hultine et al. 2004).

In the Southern (Tamaulipas) Texas Plains (EPA Level IV Ecoregion), woody encroachment in the last century has dramatically altered the vegetation structure, ecosystem processes, and suitability for livestock grazing, wildlife and other characteristics of this region. The region that was a predominantly open savanna type landscape is now a subtropical thorn shrubland comprised of dense thickets and small trees (Archer 1995). This ecosystem shift, known as woody encroachment, is often associated with increased erosion and declines in forage productivity, biodiversity and socioeconomic potential (Huxman et al. 2005). It is a common cultural practice to remove the woody vegetation of encroached ecosystems in an attempt to increase grasses and forbs. One assumption about this cultural practice is that removing woody vegetation will also decrease soil water loss through transpiration.

Woody plant removal has been taken at the national, state and local levels in an effort to mitigate the perceived negative impacts of woody encroachment. At the national level the Conservation Reserve Program, the Environmental Quality Initiative Program, and the Grazing Land Conservation initiative have provided public funds to private landowners for improving ecosystem services (Olenick et al. 2005). At the state level, programs in Texas have included the provision of State funds to clear woody plants aimed at increasing water supply (Olenick et al. 2005). At the local level, the Wintergarden Groundwater Conservation District has funded research to understand influences on potential recharge resulting from the removal of woody vegetation.

The subtropical thorn shrubland of the Rio Grande Plains is an important and extensive type of ecosystem. In North America, the area that has been converted to shrubland or woodland by woody encroachment is estimated as high as 60 million hectares (Grover and Musick 1990). In Texas, mesquite woodland covers an estimated 22 million hectares (Scifres 1980). Shrubland or woodlands are often the transitional ecosystem between moist forest and savannas. Shrublands can result from either woody encroachment into grasslands or degradation of forest. In spite of the their extensive global acreage, and geographic distribution, ecological research of the subtropical thorn shrubland ecosystem is limited relative to that of tropical savannas and tropical forests (Archer 1995).

In the northern part of the Rio Grande Plains there are 16 soil mapping units (NRCS, http://soils.usds.gove/MLRAExplorer, 10/24/2012). This soil variability might contribute to the uncertainty in estimating soil water storage and deep drainage with wood vegetation removal. To more accurately understand the role of woody vegetation on soil water conditions across the region it is necessary to study the effects across a range of soil textures. Research done in the Rio Grande Plains has been primarily conducted at a Texas A & M Agriculture and Extension Service La Copita Research

Area (LCRA). The LCRA is an 1103 ha site situated in the eastern Rio Grande Plains of the Tamaulipan Biotic Province. Although there is some variation in soil, about 80% of that research site is comprised of sandy loam uplands (Archer 1995), resulting in much research on this question to be limited to one soil type.

The existing research on woody encroached ecosystems of Texas and similar ecosystems suggests that conversion from woody vegetation to grassland and vice versa would have limited effects on groundwater recharge and quantities of water moving through the soil profile (Heitschmidt and Dowhower 1991, Weltz and Blackburn 1995, Wilcox et al. 2006b). This is based on the findings that evapotranspiration rates of shrubs and grass interspaces were similar, and surface runoff, as well as deep drainage (> 2 m), were found to be significantly greater in bare spaces than in shrub clusters and grass interspaces (Weltz and Blackburn 1995). The differing roles of deep rooted woody plants and herbaceous plants on soil water content and movement may be more complicated than suggested by evapotranspiration measurements alone. Deep rooted plants could redistribute water to deeper soil layers (hydraulic descent) and away from evaporation and shallow rooted competitors (Hultine et al. 2004). Specifically, Hultine et al. (2004) found that where *Prosopis* was present, rates of reverse flow were as high as 9 L/night following 50 and 35 mm rain events, and non-trivial amounts of water increases in deep soil layers were measured in spite of no evidence of direct recharge. Burgess et al. (2001) measured an increase in soil water of 2.3% at soil depths of 170-270 cm, they attributed the increase to the movement of water down the taproot. This suggests that deep rooted woody species can facilitate water movement into the soil. The process of hydraulic redistribution by *Prosopis* can involve large quantities of water passively moving from higher to lower root water potential but more research is needed to test this hypothesis at the ecosystem level (Caldwell et al. 1998).

Stable isotopes have been instrumental in proving the existence of hydraulic lift and hydraulic redistribution. The first research using water stable isotopes to demonstrate hydraulic lift was conducted by Dawson (1993) in which *Acer saccharum* was accessing groundwater, releasing it into shallow soil layers and it was then taken up by shallow rooted grasses. Interestingly at times where a plant seems to be accessing stream water they may actually be utilizing surface water as in the case shown by Dawson & Ehleringer (1991). In other cases, it was determined based on stable isotopes and water budget calculations that water was being moved into deeper soil layers from wetter shallow layers (Burgess et al. 1998). Stable isotopes have been used extensively to show depth of water acquisition by single species as well as partitioning of water by coexisting species in the sagebrush of California, the shortgrass steppe of northeastern Colorado and in semi-arid grasslands of southeastern Arizona and other ecosystems (Weltzin and McPherson 1997, Dodd et al. 1998, Darrouzet-Nardi et al. 2006).

This study was designed to examine how differences in soil texture affect plant water use and soil water movement in the recharge zone of the Carrizo-Wilcox aquifer. The sites were selected to be representative of the region based on varied management history of vegetation while controlling for the influence of lateral soil heterogeneity within these areas. The scale of the project is sufficiently large to capture stand level responses of soil water content as a result of soil type and vegetation. The objective of this research is to assess the influence of hydraulic redistribution on soil water contents across soil types in a subtropical shrubland.

Material and methods

Study site and vegetation

The research site is located in the Major Land Resource Area known as the northern Rio Grande Plain (28° 56’ 40” N, 100° 3’ 58” W) (NRCS, USDA). Importantly, the research site is also located on the recharge zone of the Carrizo-Wilcox aquifer (Figure 3.1). The plain is nearly level with smooth, gently rolling hills and valleys. Elevation ranges from 60 m in the southeast to 300 m in the northwest. Average annual precipitation ranges from 533 to 939 mm, decreasing from east to west across the region. The average annual temperature ranges from 19 to 22 °C.

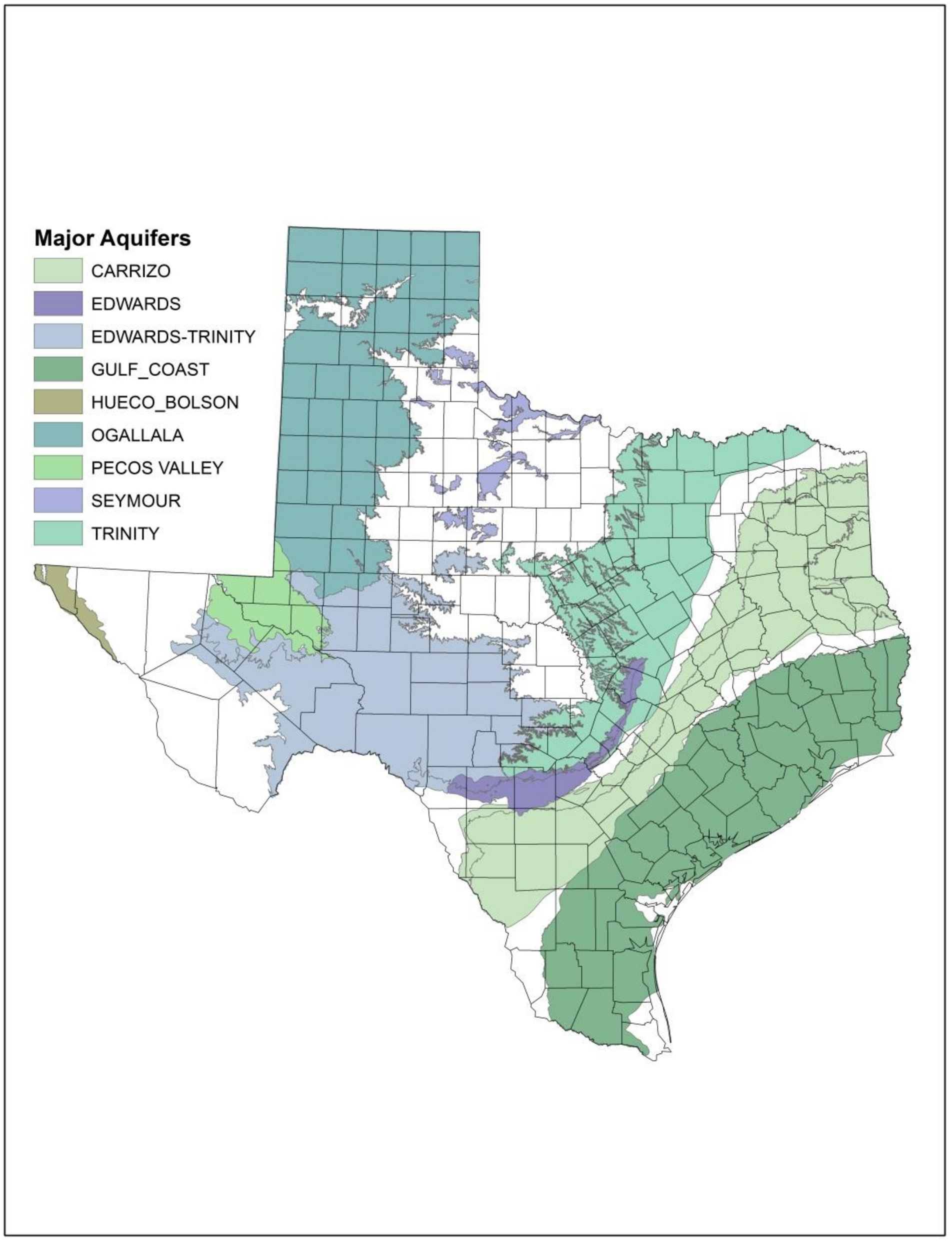

Figure 3.1 Location of research site on the recharge zone of the Carrizo-Wilcox aquifer (Texas Water Development Board, 2013).

The potential vegetation of southern Texas and northern Mexico has been classified as Prosopis Acacia savanna (Kuchler 1964). Currently, this landscape is dominated by dense woodlands, while some landscapes remain as savannas, others are thought to be still developing into closed-canopy woodlands (Brown and Archer 1990). The variability of soil characteristics interact with rooting patterns and result in a varied abundance and distribution of life-forms across these landscapes (Brown and Archer 1990). Some of the common tree and shrub species of the area are *Prosopis glandulosa* Torr. var. *glandulosa*, *Acacia berlandieri* Benth., *Acacia rigidula* Benth., *Acacia schaffneri* (S. Wats) Herm., *Diospyros texana* Scheele, *Aloysia gratissima* (Gill. & Hook.) Troncoso, *Guaiacum angustifolium*, *Acacia gregii*, *Celtis pallida* Torr., *Zizyphus obtusifolia* (T. & G) Gray var. *obtusifolia*, and *Condalia hookeri*. Grasses common to the area include: *Bouteloua curtipendula*, *Heteropogon contortus*, *Pennisetum ciliare*, *Tridens eragrostoides* (Vasey & Scribn.) Nash, *Trichloris pluriflora* Fourn., Digitari *californica* (Benth.) Henr.

Site description

The finest textured soils studied are the Chacon clay loam. Vegetation on the clay loam sites is generally very dense closed canopy with minimal understory. The intermediate texture soil is Webb series, a sandy loam. Vegetation on the Webb soil is approximately 30% cover of woody vegetation, 30% herbaceous and 40% bare ground (personal observation). The sandy soils are the Antosa-Bobillo association. These soils generally exhibit a more savanna like vegetation with large statured mesquite (> 10 ft tall), little bare ground, and up to 80% herbaceous cover. Sandy soils exhibit the greatest variation in vegetation cover, both in cover and species composition (personal observation).

Experimental design

This overall experimental design follows a randomized block full-factorial design (see Chapter II). There are three treatments applied to three soil textures for a total of nine treatment combinations. These combinations are replicated three times in separate pastures (P1, P2 & P3) with different management histories for a total of 27 plots. Each plot is approximately 0.10 hectare. Treatments are randomly assigned. The soils chosen represent the range of textures present in the recharge zone of the southern Carrizo-Wilcox aquifer. These soils were located and selected based on NRCS soil survey maps. Soil descriptions, treatments and associated vegetation are described in more detail in Chapter II. The focus here is on the control plots only to assess the impact adult woody species may have on soil moisture movement, specifically through hydraulic redistribution.

Data collection

Weather and precipitation measurements. A weather station was erected in P1 (Figure 2.1). The weather station was equipped to record precipitation using a TE525 tipping bucket rain gauge (Texas Instruments, Dallas, TX, USA), photosynthetically active radiation using a LI-190SB Quantum sensor (Li-Cor, Lincoln, NE, USA), temperature and humidity using a HMP45C temperature and relative humidity probe (Campbell Scientific, Logan, Utah, USA), and wind speed and direction using a 03001 R.M. Young Wind Sentry Anemometer and Vane (Campbell Scientific, Logan, Utah, USA), powered by a 12v battery charged by a solar panel and data collected and stored on a CR10X data logger (Campbell Scientific, Logan, Utah, USA). Data was logged hourly, and downloaded and monitored monthly. To account for small-scale variability in rainfall, two additional TE525 tipping bucket rain gauges were installed at P2 and P3. These collected rainfall and temperature measurements hourly using Hobo UA-003-64 data loggers (Onset, Cape Cod, MA, USA) and were monitored and downloaded monthly.

Volumetric water content measurements. Soil water content was measured approximately monthly in all control plots. A total of 18 soil profiles (two within each plot) were sampled to a depth of 180 cm using a model 503DR, neutron moisture meter (NMM) (Campbell Pacific Nuclear, Martinez, CA, USA). Measurements were taken in 20 cm increments to a depth of 180 cm. The values obtained from the NMM were applied to a linear regression calculated from in situ calibration measurements to obtain volumetric soil water content (VWC, cm^3 H_2O/ cm^3 soil). The calibration for the NMM was conducted in the field. Appendix A contains additional details of calibration. The fitted calibration curve is described by the following equation: VWC (cm^3 H_2O *cm^{-3} soil) = 0.1804 x - 0.0986, where x = Count ratio (count/standard).

Isotope sampling. Precipitation was collected on a monthly basis from each of the three pastures and analyzed for oxygen and hydrogen isotope ratios ($\delta^{18}O$ and δ^2H, ‰). Two cm of mineral oil was poured into gallon size Nalgene bottles to prevent evaporation of precipitation in the field prior to collection. The bottles were then fitted with funnels and secured in the field. The precipitation amounts and isotope values were combined for the three locations and presented as a single weighted average.

Soil samples were collected for soil water isotope analysis. Samples were collected for all three soil textures in two pastures for control plots during each sampling period. Repeat sampling was conducted in summer 2011, early summer 2012 and late summer 2012 (Table 3.1).

Repeat samples were collected within 5 m of previous samples for each plot. Samples were collected at depth intervals of 18-23, 38-43, 58-63, 88-93, 118-123, 158-163 and 198-203 cm. Holes were drilled in the soil using a Giddings probe with a bucket auger. Caution was taken to not heat the soil during sampling. A plug of soil was removed from the bottom of the drilled hole. Soil samples were immediately placed into scintillation vials, wrapped in parafilm and placed in a cooler in the field. Samples were placed into a freezer each day and were kept frozen until extraction. Suberized stem samples of *Prosopis glandulosa* were collected in control plots at the same time as soil samples.

Table 3.1 Stable isotope sampling dates and locations for stems and soils. * stems collected 7/28/2011 ** no stem or soil sample collected

Location, Soil, Treatment	Summer 2011	Early Summer 2012	Late Summer 2012
P3, sand, control	7/17/2011-7/18/2011*	5/22/2012	7/25/2012
P2, sand, control	7/1/2011-7/2/2011	5/24/2012	7/26/2012
P1, clay loam, control	6/28/2011	5/23/2012	7/26/2012
P2, clay loam, control	9/3/2011	6/3/2012	**
P3, clay loam, control	7/24/2011*	5/23/2012	7/24/2012
P3, loam, control	7/15/2011	6/3/2012	7/24/2012
P2, loam, control	**	6/4/2012	7/26/2012

Lab analysis. Soil and stem water were extracted in the lab using cryogenic vacuum distillation (West et al. 2006). Isotopic analysis was performed at the SIBS Lab (Texas A&M University, College Station). Microliter quantities of water were injected directly into a high temperature conversion / elemental analyzer (TC/EA) coupled to a Delta V isotope ratio mass spectrometer (Thermo Scientific Inc., Waltham, MA). Hydrogen and oxygen isotope ratios were both obtained from the analysis. Isotope ratios are expressed in standard delta notation in ‰ as:

$$\delta^{N}E = (R_{sample} / R_{standard} - 1) \times 1000$$

where N is the heavy isotope of element E, and R is the ratio of heavy to light isotopes ($^2H/H$ or $^{18}O/^{16}O$) of the unknown sample (R_{sample}) and an international standard ($R_{standard}$). The standard for water is V-SMOW (Vienna Standard Mean Ocean Water).

Results

During the period of data collections, the region experienced an exceptional drought. The 1st summer following vegetation removal treatment (summer 2011) was very dry and there was no herbaceous recovery on the treated plots, and no herbaceous growth in the control plots. Fall rain resulted in some grass recovery in November 2011, but overall herbaceous recovery was minimal. This allowed for summer 2011 measurements to be analyzed as effectively woody component only. The winter precipitation patterns of 2011-2012 were typical, but soil water content was at a deficit, so we do not expect the results to be as they would under relatively normal conditions the previous summer.

There were two time periods of interest for analyzing the composition and amount of precipitation. Precipitation occurring during cooler months is generally expected to move into deep soil horizons because little is lost by transpiration from winter deciduous plants, and evaporative demand is low. This is the time period when we expect to see recharging of deep soil layers. Due to kinetic effects on water stable isotope values, we also expect cool season precipitation to have a relatively more negative $\delta^{18}O$ value. The total rainfall amount for winter of 2011-2012 from October thru February was 221 mm (Figure 3.2). The $\delta^{18}O$ values are -3.9‰, -4.7‰ and -4.8‰ for October, November/ December and January/February, respectively (Figure 3.3). Limited access to the field and samples required the combining of isotope values for November/December and January/February. The $\delta^{18}O$ weighted average of precipitation from October thru February is -4.45‰.

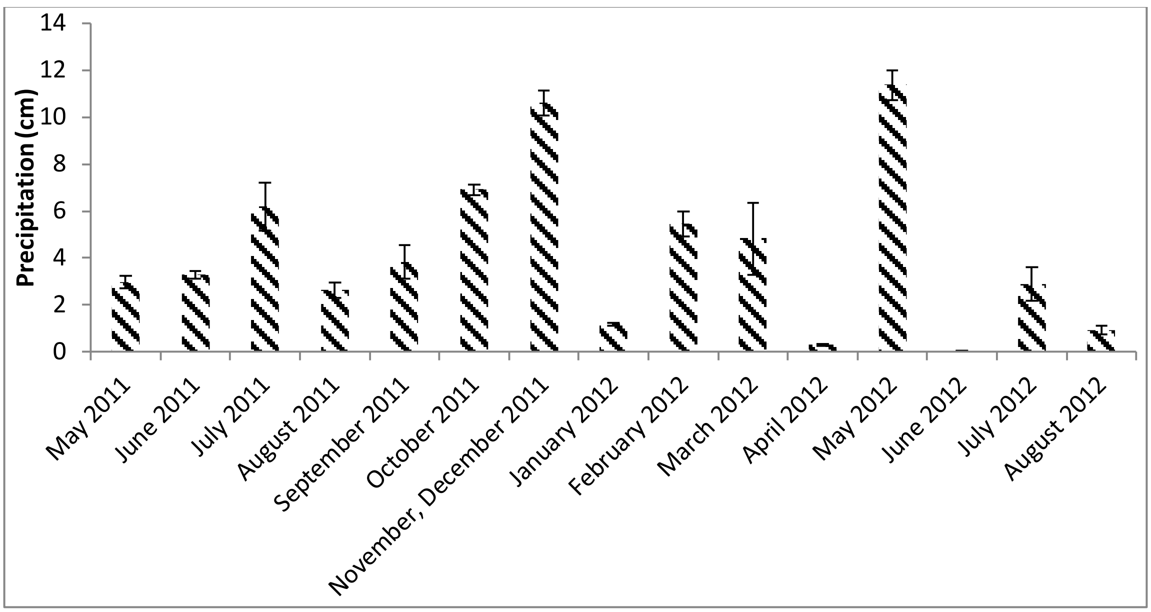

Figure 3.2 Monthly precipitation averaged among the three sites P1, P2 and P3. Error bars are standard deviation and represent variation between pastures.

Precipitation from March thru May of 2012 totaled 165 mm. Stable isotope $\delta^{18}O$ values for March, April and May were -2.7‰, -2.2 ‰ and -2.2‰ respectively (Figure 3.3). The precipitation from March would likely have had time to be lost by ET due to the flush of herbaceous vegetation in early March. If the water was not lost, it should have been isotopically enriched by evaporation prior to the early summer 2012 sampling campaign that began in mid-May 2012. April precipitation amounts were negligible. Early summer 2012 samples were collected within 2 weeks of the 114 mm of rain in May. Soil water contents were lowest for 2012 just prior to May precipitation, and undetectable in the sandy soils to 60 cm depth (Figure 3.9). The volume of water from the May 2012 precipitation event was enough to have overwhelmed the isotopic signature of evaporatively-enriched water in the shallow soil profile prior to sampling. For these reasons, we expect surface soil water isotope values in the early summer 2012 soil samples to resemble May 2012 stable isotope precipitation values.

The time between early 2012 and late summer 2012 soil isotope sampling received 28mm of rain with a $\delta^{18}O$ value of -1.9‰, which is less negative than the heavy rainfall event in May. For this reason, we expect late summer stable isotope values to reflect May 2012 precipitation altered by evaporation. The effects of evaporation would cause the $\delta^{18}O$ values to become less negative.

In the each of the control plots the $\delta^{18}O$ (‰) values for the three sampling periods are more negative deeper in the soil profile and increase exponentially in the shallow soils (Figure 3.8). The similar exponential curves across time periods in soil water response is expected due to no manipulation of vegetation prior to sampling. In spite of the isotopic differences between winter and summer precipitation, the isotope profile show no distinction between winter, summer and evaporatively enriched waters even shortly after a significant rain event (Figure 3.3).

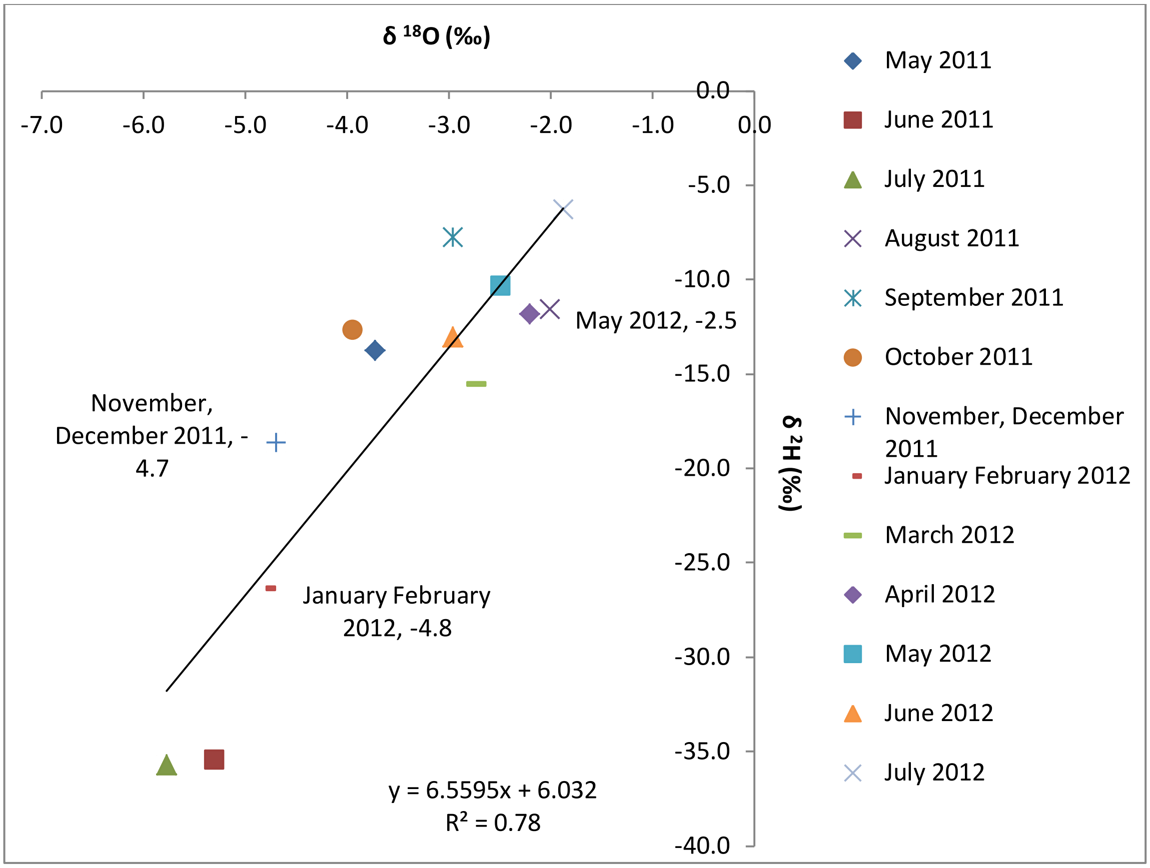

Figure 3.3 Precipitation stable isotope values weighted for three pastures.

Stable isotope values from the clay loam soil water become more negative from 40 to 200 cm between the early summer 2012 and late summer 2012 sampling periods (Figure 3.4). These values are averages of four samples from two replicate plots (four samples total). In early summer 2012, $\delta^{18}O$ values from 40 to 200 cm ranged from -2.06 to -3.10‰. In late summer 2012 $\delta^{18}O$ values from 40 to 200 cm ranged from -2.52 to -3.73‰. Volumetric water content was measured on 6/1/2012 and 7/27/2012 very near the time when soil and stem isotope samples were collected in early and late summer 2012. Between those two dates, there was a decrease in soil water at 0 to 80 cm deep and an increase in soil water from 100 to 180 cm deep in the clay loam soil (Figure 3.5). Soil water on 6/1/2012 at 0 to 80 cm deep ranged from 0.137 to 0.167 m^3 m^{-3}. Soil water on 6/1/2012 at 100 to 180 cm deep ranged from 0.147 to 0.170 m^3 m^{-3}. Soil water on 7/27/2012 at 0 to 80 cm deep ranged from 0.118 to 0.160 m^3m^{-3}. Soil water on 7/27/2012 at 100 to 180 cm ranged from 0.150 to 0.174 m^3m^{-3}.

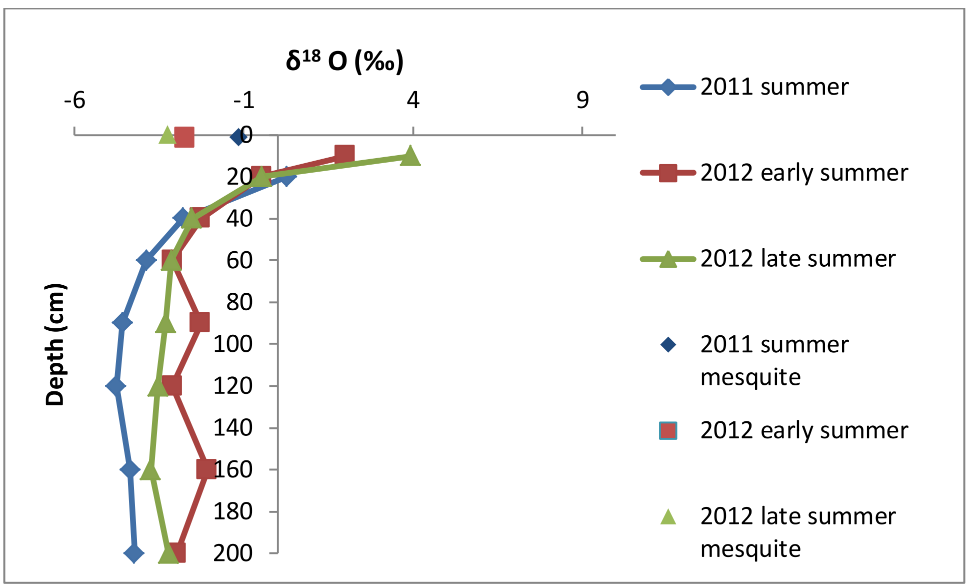

Figure 3.4 Depth vs. δ^{18}O for clay loam soil control plots. *Prosopis* stem water is plotted on y-axis.

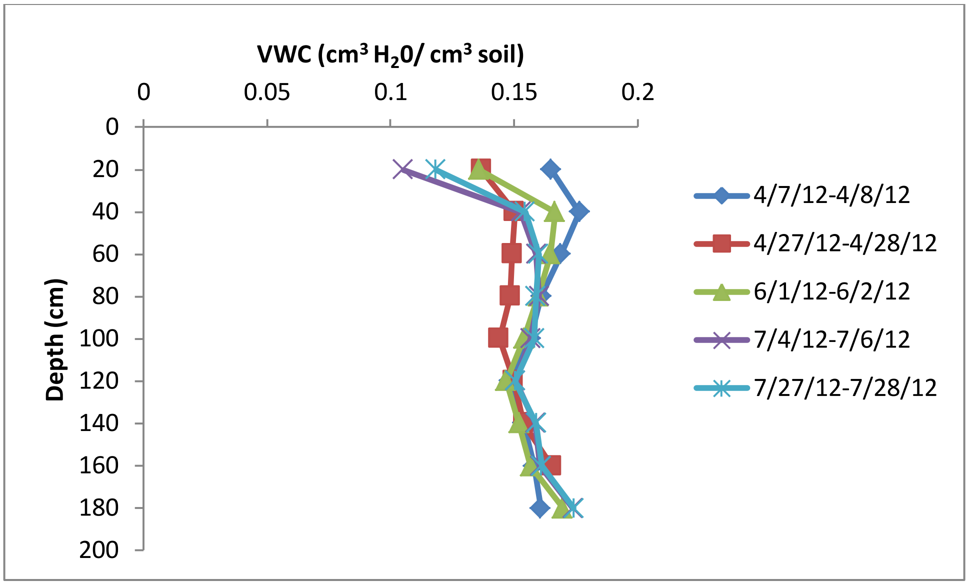

Figure 3.5 Soil water content for clay loam control plots.

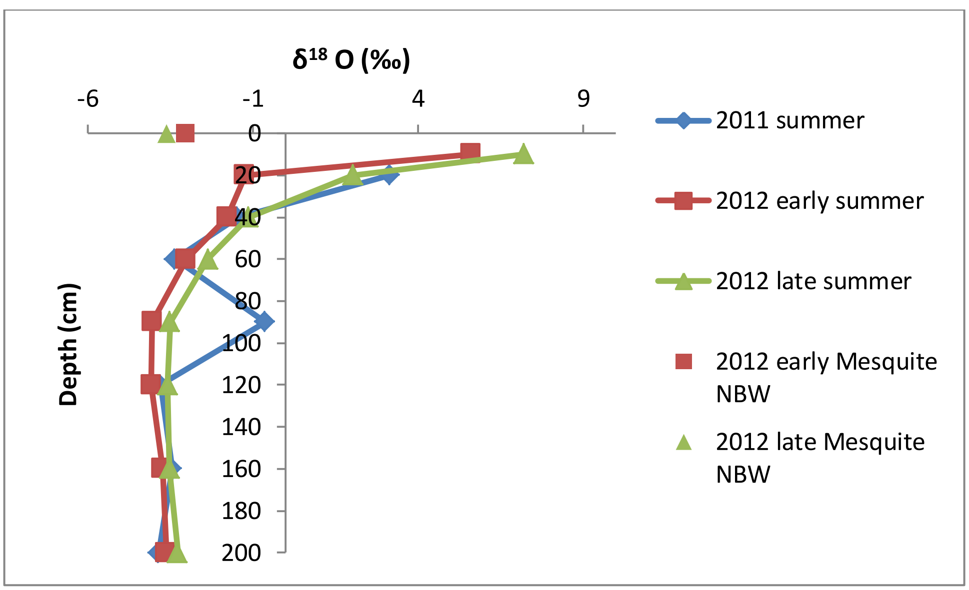

Figure 3.6 Depth vs. $\delta^{18}O$ for loam soil control plots. *Prosopis* stem water is plotted on y-axis. *Prosopis* stem water was not collected in 2011 for loam soil.

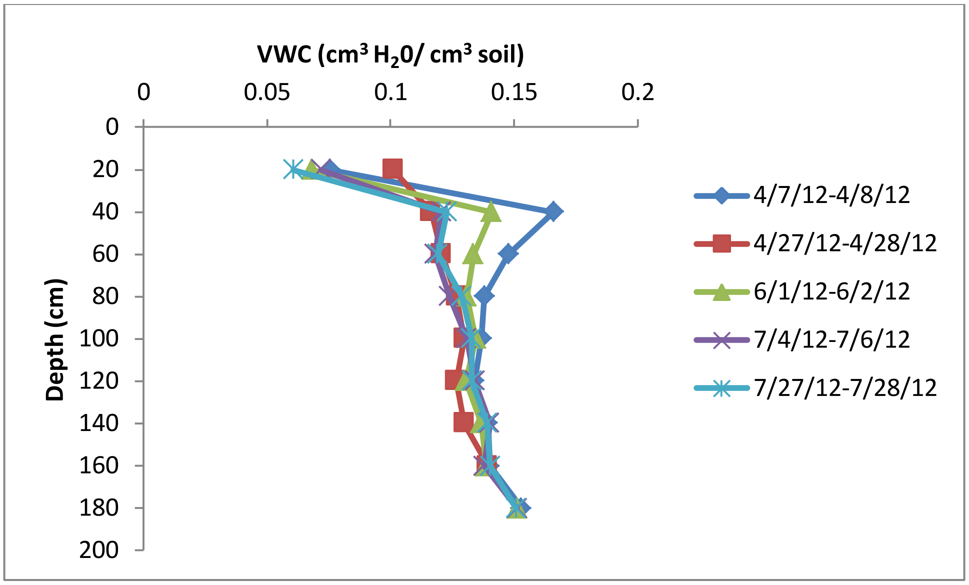

Figure 3.7 Soil water content for loam control plots averaged from three replicates.

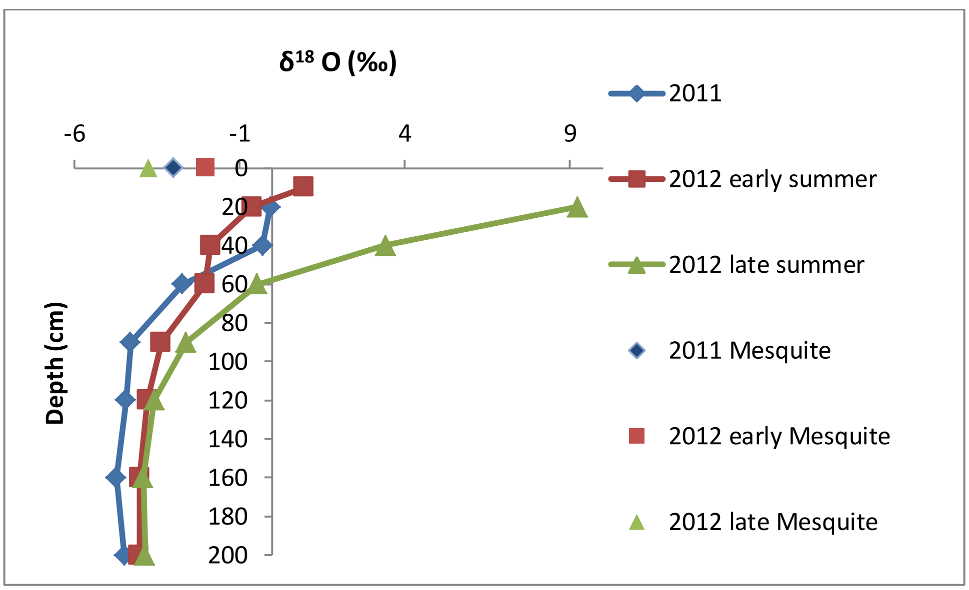

Figure 3.8 Depth vs. $\delta^{18}O$ for sandy soil control plots. *Prosopis* stem water is plotted on y-axis.

Loam soils stable isotope values become less negative across the entire profile between early and late summer 2012 sampling periods (Figure 3.6). The $\delta^{18}O$ values in early summer 2012 ranged from -1.24 to 5.61‰. In late summer 2012 the $\delta^{18}O$ values ranged from -1.13 to 7.2‰.

Soil water content for sandy control plots shows very little variation at all depths throughout the sampling period as well as around the time of isotope sampling (Figure 3.9). Soil water at 20 & 40 cm remain below 0.05 m^3m^{-3} from August 2011 through September 2012. Soil water at 60 cm only exceeds 0.05 m^3m^{-3} in January, April and June 2012. The greatest difference between soil water at depths of 80 to 180 cm ranged from 0.010 to 0.018 m^3m^{-3} for the entire sampling period.

Measurements were taken monthly. During very wet periods access was limited by road conditions imposed by clay soils. Road conditions limited access up to a week and a half following heavy rains in spring or summer. In winter, the soils dried more slowly, but did not limit access for greater than three weeks. The limitation was equal across all soil types and treatments, but it could mean that observations were biased somewhat to drier conditions.

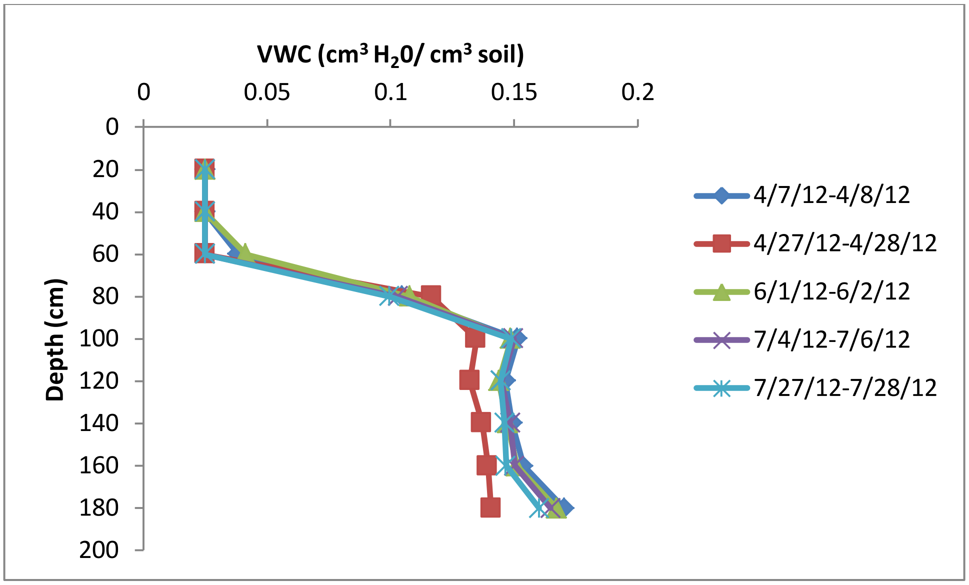

Figure 3.9 Soil water content for sandy control plots averaged from three replicates.

Discussion

When considering managing rangeland for maximum water yield, the complex interaction of vegetation, soil characteristics, and the response of soil water movement must be taken into account. In particular, it should be recognized that vegetation may not only reduce soil moisture through transpiration, but also has the potential to affect its vertical and horizontal distribution in soils either directly through hydraulic redistribution or indirectly through the formation of root channels The results presented here provide support for the hypothesis that hydraulic redistribution is occurring in the shrubland of the northern Rio Grande Plains. This hypothesis is consistent with earlier speculation in a similar system (Midwood et al. 1998). I will use soil water measurements as well as stable isotope data to make inferences about below ground processes. For ease of discussion, I will focus on the stable water isotope measurement of $\delta^{18}O$. The trends discussed here are based on an empirical analysis of the graphs presented. Statistically there was no difference in the early and late summer 2012 $\delta^{18}O$ values based on a paired Student's T test. The focus of this section will be control plots on the three soil types studied: clay loam, sandy loam and sand.

The hypothesis that hydraulic redistribution is occurring is supported for all three soils studied. The strongest support is the change in $\delta^{18}O$ values in the clay loam profile. The stable isotope values become more negative between early summer 2012 and late summer 2012. If the water was being affected by evaporation during this time period, the isotope values should have become less negative. There was also an increase in soil water at 100, 120, 140, 160, & 180 cm. Water in the shallow soil should have been similar to May rain with a $\delta^{18}O$ value of -2.2 ‰, or slightly less negative from mixing with evaporatively enriched shallow soil water. If water was moving down in the profile from the May rain, isotope values would become less negative instead of more negative. Between sampling periods there was 28 mm of rain that fell, and that rain had a $\delta^{18}O$ value of -1.9‰. The rain in between summer 2012 sampling periods is less negative than the May rain that moved into the profile prior to early summer sampling, and does not explain the shift in soil isotope values to more negative values. The most plausible explanation for the shift in stable isotope values in the clay loam soil profile is that water is being accessed at depths deeper than 2 m and hydraulically lifted into the 40 to 200 cm depth of the soil profile.

There is evidence within the clay loam profile of water movement into the soil being facilitated by woody vegetation. For hydraulic redistribution to occur, there only needs to be difference in water potential and roots to maintain connectivity with the soil. So, in a soil with little textural or structural difference, water will move from wetter layers to drier soils due to the potential gradient, regardless of super position. The early summer 2012 sampling was conducted shortly after 113 mm of rain fell. At depths of 90 and 160 cm in the soil profile, soil water resembles the $\delta^{18}O$ value of May (-2.2‰). The soil water isotopes at 120 and 200 are more negative than the rain that fell in May. If water was moving into the profile through the soil matrix, it is unlikely that only certain depths would resemble the most recent rain event, especially considering this soil is fairly homogenous in texture and water content. Later in the summer, there is a larger shift in $\delta^{18}O$ values at 90 and 160. The shift from early to late summer is smaller at 120 and 200. If depths of 90 and 160 are hydraulically connected by the roots, it is reasonable to think that when shallow soils are wet, water is moved to deeper drier layers (90 and 160 cm). If those depths are strongly connected to the plant via roots, they should also become drier in the daytime, and be recharged at night by deeper water being lifted. It is also reasonable to think that once shallow soils dry during summer, that water is accessed from beyond 2 m and lifted to the depths of strong connectivity. This further supports that depths of 90 and 160 are hydraulically connected thru the root system.

Additional support for hydraulic redistribution in the clay loam soils is decrease in the range of isotope values from 60-200cm between early and late summer 2012. In other words, there was a smoothing of the isotope profile as a result of mixing in spite of little change in amount. The early summer profile has a larger range of isotope values. Isotope values becoming less negative over time in depths less than 60 cm reflects the effects of evaporation on $\delta^{18}O$ values. Isotope values becoming more negative deeper in the soil profile reflect mixing of water at 80 to 180 cm with water from deeper than 180 cm that has a more negative isotopic value. Movement downward of water from shallow to deeper soils would cause $\delta^{18}O$ values to become less negative, which was not observed. The shift in isotope values in combination with the increase in soil water content further supports the occurrence of hydraulic redistribution. This does not reflect a change by gravitational flow.

Sandy loam soils exhibit some evidence of hydraulic redistribution. The effects are not as pronounced as in the clay loam soils. This finding is consistent with the water potential gradient being greater in clay loam and less in sandy loams, a basic requirement for hydraulic redistribution to occur. In the sandy loam, isotope values become less negative than precipitation inputs to 40 cm soil depth between early and late summer 2012 sampling periods. This shift indicates loss of water by evaporation. The loss of water by evaporation at 10 cm is less at 20 cm soil depth. This could be due to the small input of rain between sampling periods. This could also be support for hydraulic redistribution where the effects of evaporation are being diluted by the addition of water with a more negative isotope value from deeper in the soil, although we cannot separate these two mechanisms here. Isotope values are less negative in a relatively equal magnitude from 60 to 200 cm soil depth. It is unlikely that the effects of evaporation are equal between 60 and 200 cm. Comparison of δ^2H to $\delta^{18}O$ to the local meteoric water shows no evaporative enrichment of waters at or below 90 cm. The shift in $\delta^{18}O$ values below 90 cm could be attributed to the mixing of water from shallow to deeper soils as a result of matric flow. It is unlikely that the amount of water distributed through the deeper horizons is equal across depth, again pointing to mixing by hydraulic redistribution. After the early 2012 sampling period, soils at 80 cm or shallower became drier, and soils below 80 cm became wetter. It would be possible that water from May moved by gravity to deeper soil layers, and that is the process responsible for the isotope values becoming less negative. Because the shallow soils are very dry and deeper soils become wetter, gravitational flow cannot be excluded as a factor influencing the changes in soil water isotopes.

Sandy soils are less conclusive in regards to the effects of hydraulic redistribution. Sandy soils have a strong textural shift to a sandy clay loam at 80 cm soil depth. The clay loam horizon continues to 180 cm soil depth which is also the depth at where sampling stops. Deeper than 180 cm soil depth the soil texture is very sandy, similar to the shallow soils. Burgess et al. 2001 discusses the effect of Eucalyptus moving water into deep layers in a soil with a very similar profile. It is still very plausible that water is being moved from shallow sandy layers, through the clay loam horizon and into deep sandy layers below 180 cm by roots. This is not evidenced in soil water isotopes because sampling ends at 180 cm, but is supported by *Prosopis* stem water isotope data.

Stem water from *Prosopis* was used to help understand soil water dynamics in the shrubland ecosystem. In the sandy control plots stem water shows a shift from early to late summer 2012. The isotope values for mesquite stems become more negative in late 2012. This shift could represent the plants using some summer water with isotope values around -2‰, and deeper water with isotope values around -4.5‰. Another possibility is that the shift was to accessing water at 100-200 cm, but soil water content does not change at 100 to 200 cm soil depth between early and late 2012. It is possible plants were using only a small amount of water from 100-200 cm and that the isotopic signature could be changed by a small contribution of water from that is not reflected by a change in soil water content. Alternatively, Prosopis may be accessing water from sandy soil layers deeper than 200 cm that was redistributed from shallow sandy soil layers during times when shallow layers were wet and deeper layers had been depleted.

Most of the inferences regarding the occurrence of hydraulic redistribution in the shrubland ecosystem are based on observations of changes in soil water stable isotopes. Some of those inferences were bolstered by stable isotope data from *Prosopis glandulosa*. There are numerous other shrub species present in the control plots, and should not be disregarded in their roles in soil water dynamics. For this reason it is important to consider these findings in the context of the ecosystem, and not strictly in regards to *Prosopis glandulosa*. *Prosopis* was chosen because it is believed to be deeply rooted at times, and is common in all of the soil types present.

There are additional aspects of the data that are notable that neither support nor deny the occurrence of hydraulic redistribution. For all soils studied, the *Prosopis* stem

$\delta^{18}O$ values are more negative in late summer 2012 than in early summer 2012. This indicates a shift in the depth of water acquisition to deeper water during the later drier part of the summer. Within the sandy loam soils the stem isotope values become more negative, yet the entire soil profile to 2 m becomes less negative. Again, suggesting that the depth of water acquisition is becoming deeper as the summer progresses. All of the control plots for each soil type show a consistency in regards to the amount of loss of water by evaporation based on changes in soil water isotope values and soil water content. It is difficult to use these values to determine the amount of water lost by evaporation because of the complications arising from hydraulic redistribution. The benefits of hydraulic redistribution were discussed by Hultine et al. (2004). They discuss briefly the implications of hydraulic redistribution on reducing water and nutrient deficits, moving water away from evaporation, and how it could be an important factor in partitioning evaporation and transpiration on daily and seasonal timescales. However, what is perhaps less appreciated is the potential role of hydraulic redistribution in obscuring interpretation on soil moisture stable isotope profiles (Allison et al. 1983, Clark and Fritz 1997).

Limitations

Access to (Burgess et al. 1998, Burgess et al. 2001, Hultine et al. 2004, Zou et al. 2005) plots in remote locations was limited at times due to road conditions. This limitation affected our ability to take NMM measurements immediately following rain events. This limitation restricted our ability to measure soil water content when sandy soils were wet, because roads were impassible due to clay soils. This skewed the data to appear as if sandy soils were very dry at all times when in fact we were just not able to access them when they were wet. This limitation however is thought to only affect shallow soil measurements, not deep soil measurements. The effects on deeper soils in the sandy plots is not expected to be as time sensitive due to the increase of clay content around 80 cm, and the subsequent decrease in hydraulic conductivity. Based on saturated hydraulic conductivity (K_{sat}) estimates for Antosa fine sand, the maximum rate of movement through the sandy horizon would be 1.98 in hr^{-1}, and the most limiting horizon, which begins at 80 cm has a minimum rate of 0.20 in hr^{-1}. This equates to a maximum of 0.6 days for water to move under saturated conditions through 80 cm of sand and 8.2 days to move through 1 m of sandy clay.

Contributions and future research

The occurrence of hydraulic redistribution has been shown in many ecosystems (Burgess et al. 1998, Burgess et al. 2001, Hultine et al. 2004, Zou et al. 2005). The importance of hydraulic redistribution, and benefit to individual species has yet to be shown. This process appears to play a strong role in the distribution of water within a soil profile. It is important to understand how plants are affecting soil water, especially if there is an intention to manage woody vegetation for maximum water yield.

Hydraulic redistribution has been shown in a number of other woody species (Burgess et al. 1998, Ludwig et al. 2003, Hultine et al. 2004). Among different species of *Prosopis* hydraulic redistribution have been shown (Caldwell et al. 1998, Hultine et al. 2004, Zou et al. 2005). The results of this study show that at the ecosystem level, hydraulic redistribution by woody plants is a contributing factor in soil water distributions. This process could be influencing the distribution of woody and herbaceous vegetation in the northern Rio Grande Plains.

The most important implication of the occurrence of hydraulic redistribution in the northern Rio Grande Plains is that the presence of woody plants has the potential to increase the amount of water moving into deep soil horizons. Requirement for the occurrence of hydraulic redistribution are occasional dry soils and maintenance of contact and conductivity with surrounding soil (Caldwell et al. 1998, Hultine et al. 2004). The removal of woody vegetation was shown here to reduce the amount of water in the soil profile in association with roller chopping in clay loam soils. These results are counter to the desired effect of increasing deep drainage by the removal of the woody vegetation.

When considering managing rangeland for maximum water yield, the complex interaction of plant soil and water must be taken into account. Shifts from grassland to shrublands influence on water use are more dynamic than just differences in transpiration and depth of acquisition. The results from this study show that these effects are soil dependent and can be variable within an ecosystem. Before moving forward with removal of woody vegetation to increase groundwater recharge more work is needed to understand the interaction of other woody species on soil water dynamics. In this study we focused on *Prosopis*, but there are abundant other species present such as acacias and evergreen species that need to be considered in regards to their role in plant soil water dynamics of this ecosystem.

Some areas need further investigation to better understand the role woody vegetation plays on hydrology in the Rio Grande Plains. Specifically, the depth of influence of evaporation need to be better understood for this system so a water balance approach could be more accurately applied. The varying effects of herbaceous and woody vegetation on shallow soil water dynamics increase the complexity associated with understanding loss of water by evaporation in this system, as highlighted by stable isotope data. Another area needing more research is the determination of depth of acquisition of water by woody species in the area. Quantification of the amount of water used from depths beyond the majority of roots is important to understanding the effects of woody vegetation on the soil water dynamics and the potential to increase deep drainage. Assuming hydraulic redistribution in occurring, the persistence of roots following the removal of woody vegetation could add to the understanding of lasting effects of roots as conduits as well as questions regarding root longevity.

Conclusions

The evidence for hydraulic redistribution occurring by *Prosopis glandulosa* is strong for clay loam soils in the northern Rio Grande Plains. The shift in $\delta^{18}O$ in soils and stems is most easily explained by hydraulic redistribution, and less by gravitational or capillary flow. Clay loam soils can exhibit very negative water potentials and *Prosopis* is known to withstand very negative water potentials. The extent to which hydraulic redistribution occurs is strongly soil dependent, and appears more prevalent in soils with higher clay content. Connectivity with the soils and roots is required for hydraulic redistribution. Small pore sizes in clay loam soils may help to promote the maintenance of conductivity furthering the likelihood of hydraulic redistribution occurring in fine textured soils. High temperatures and evaporative demand for extended periods of time cause soils to become very dry. When shallow soils are wetted the difference in water potential can be great between shallow and deeper soils. This promotes the movement of water by roots acting as conduits into drier soil horizons.

The data discussed in this chapter show the influence of hydraulic redistribution in the mesquite shrubland of the northern Rio Grande Plains. Movement of water into the soil profile by roots strongly affects the soil water balance. This should be considered when managing rangelands for maximum water yield. The impacts of roots in the soil profile following woody vegetation removal need to be further considered. The use of stable isotopes adds to understanding of water pathways. Further research needs to be conducted on soil water dynamics and the role of residual roots on short and long term timescales following shrub removal.

CHAPTER IV

CONCLUSIONS

The average rooting depth of the northern Rio Grande plains ecosystem following woody encroachment is approximately 150 cm across a range of soil textures. Although there may be cases where roots are accessing water at deeper depths, the major root function in our system is occurring in soils less than 120 cm deep. The combination of low average annual precipitation, soil texture, structure and vegetation on soil water movement limit the likelihood of deep drainage occurring in the northern Rio Grande Plains. For deep drainage to occur, soils must be saturated beyond the depth of roots. When water is present in the root zone, it is quickly utilized by plants and rarely has time to move beyond 120 cm. Water that is moved deep into the soil profile by woody vegetation is later utilized by deep rooted woody species.

The evidence for hydraulic redistribution occurring by *Prosopis glandulosa* is strong for clay loam soils in the northern Rio Grande Plains of s. Texas. The shift in $\delta^{18}O$ in soils and stems is most easily explained by hydraulic redistribution, and less by gravitational or capillary flow. The extent to which hydraulic redistribution occurs is strongly dependent on soil texture and structure, and appears more prevalent in soils with higher clay content. Connectivity with the soils and roots is required for hydraulic redistribution. Small pore sizes in clay loam soils may help to promote the maintenance of conductivity furthering the likelihood of hydraulic redistribution occurring in fine textured soils. High temperatures and evaporative demand for extended periods of time cause soils to become very dry. When shallow soils are wetted the difference in water potential can be great between shallow and deeper soils. The strong water potential gradient promotes the movement of water by roots acting as conduits into drier soil horizons.

The influence of hydraulic redistribution in the mesquite shrubland of the Rio Grande Plains has been shown through the use of stable isotopes. Movement of water into the soil profile by roots of woody vegetation has an effect on soil water dynamics. The impacts of roots in the soil profile following woody vegetation removal needs to be considered when managing rangelands for maximum water yield. The use of stable isotopes adds to understanding of water pathways. Further research needs to be conducted on soil water dynamics and the role of residual roots on short and long term timescales following shrub removal.

The combination of roller chopping and sandy soils resulted in the greatest change in soil water content at depths of 60-120 cm. Under average rainfall of ~ 600 mm annually this combination is not expected to result in deep drainage. The combination of roller chopping and clay loam soils reduces total soil water content and the amount of water moving to deep soil horizons and beyond the root zone. The effect of roller chopping on clay loam soil has the potential to decrease the amount of water moving beyond the root zone and becoming available for recharge within two years following woody vegetation removal. Due to high evaporative and transpirational demand by woody or herbaceous plants in combination with soil physical limitations, the likelihood of diffuse recharge occurring in the “recharge” zone of the Carrizo-Wilcox aquifer is very low.

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APPENDIX A

NEUTRON PROBE CALIBRATION

The calibration for the NMM was conducted in the field. Count ratios were taken in the field, and compared to measured volumetric water content from soils collected during calibration. Soil samples were collected at depth increments of 10-30, 31-50, 51-70, 71-90, 91-110, etc. to 190 cm using a 2 in diameter bucket auger (Giddings Corp., Windsor CO, USA). Soils were oven dried in the lab to determine volumetric water content. Bulk density was determined by weighing the known volume of soil in the lab. Calibration measurements were taken on all 3 soil types.

The fitted calibration curve is described by the following equation:

Volumetric soil water content (cm^3 H_2O *cm^{-3} soil) = 0.1804 x - 0.0986, where x = Count ratio (count/standard).

Typically soils are sampled for calibration of the NMM under dry and wet conditions to capture the range of soil water contents to be measured by the neutron probe. For the calibration used here, only dry soils were sampled. Field conditions limited our ability to sample during wet conditions. The range of soil water contents measured during “dry calibration” as prescribed by the manufacturer is within the range of values measured during our sampling period. During the sampling period, most of the soils remained relatively dry. For this reason the dry calibration alone is assumed to be sufficient to predict soil water content.

Before each round of measurements a new "standard count" was determined for the moisture meter. To determine the standard count the meter was place on end on top of the lead plate located on top of the carrying case with source/detector inside the shield. The case was then placed on top of a Rubbermaid container to elevate it from the soil so the probe would not be detecting soil moisture during the determination of the new standard. New standard counts were recorded each month and used to determine the count ratio (count/standard). Accuracy of the device is monitored during the standard measurement. A chi squared value is determined to assure a consistent reading by the device. Chi square values between 0.75 and 1.25 were considered acceptable at a probability level 95% (IAEA, 2008). When the chi squared value was not within that range, the standard was redone. A plot of the standard values over time shows there was no gauge failure throughout the period of measure.

Table A.1 Soil water measurements and counts for calibration.

Soil	Depth	Ratio (x)	Avg Θ (y)	Predicted Θ	Residuals	RMSE
CKB P3	40	0.8641	0.0638	0.0573	0.0000	0.0188
	60	0.9878	0.0676	0.0796	0.0001	
	80	1.0257	0.0742	0.0864	0.0001	
	100	1.0563	0.0735	0.0920	0.0003	
	120	1.0658	0.0743	0.0937	0.0004	
	140	1.1969	0.0809	0.1173	0.0013	
CKB P1	20	0.9415	0.0756	0.0713	0.0000	
	40	1.2280	0.0974	0.1229	0.0007	
	60	1.2622	0.1175	0.1291	0.0001	
	80	1.2256	0.1401	0.1225	0.0003	

Table A.1 Continued

Soil	Depth	Ratio (x)	Avg Θ (y)	Predicted Θ	Residuals	RMSE
	100	1.2130	0.1291	0.1202	0.0001	
	120	1.2164	0.1680	0.1208	0.0022	
	140	1.3852	0.1657	0.1513	0.0002	
ABC P1	40	0.8579	0.0495	0.0562	0.0000	
	60	1.2204	0.0882	0.1216	0.0011	
	80	1.4905	0.1621	0.1703	0.0001	
	100	1.2706	0.1532	0.1306	0.0005	
	120	1.2218	0.1602	0.1218	0.0015	
	140	1.3662	0.1468	0.1479	0.0000	
ABC P3	60	1.0871	0.0895	0.0975	0.0001	
	80	1.2014	0.1170	0.1181	0.0000	
	100	1.2668	0.1217	0.1299	0.0001	
	120	1.2869	0.1370	0.1335	0.0000	
	140	1.2051	0.1311	0.1188	0.0002	
VAT P1	20	1.0182	0.0655	0.0851	0.0004	
	40	1.1584	0.0859	0.1104	0.0006	
	60	1.1617	0.1086	0.1110	0.0000	
	80	1.1702	0.1169	0.1125	0.0000	
	100	1.1198	0.1164	0.1034	0.0002	
	120	1.0930	0.1156	0.0986	0.0003	
	140	1.0775	0.1144	0.0958	0.0003	
WEB P2	60	0.8911	0.0604	0.0622	0.0000	
	80	1.0163	0.0702	0.0847	0.0002	
	100	0.8770	0.0566	0.0596	0.0000	
	120	0.8400	0.0619	0.0529	0.0001	
	140	0.8380	0.0857	0.0526	0.0011	

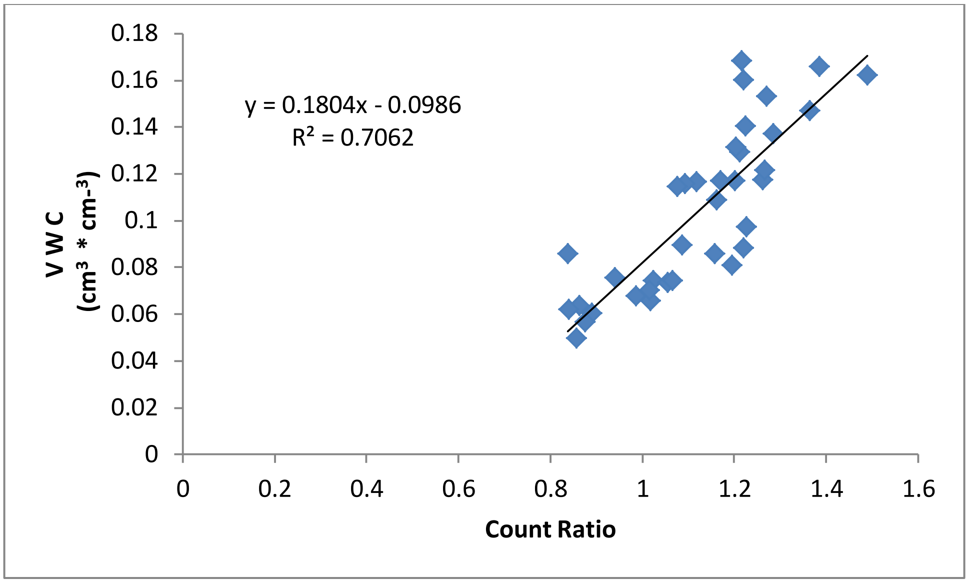

Figure A. 1. Measured volumetric water content vs. count ratio for neutron probe calibration.

APPENDIX B

MEAN SOIL WATER CONTENT

Table B. 1. ANOVA results of total soil water contents for all months.

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
7/28/11	Soil	ABC	8.633	1.817	5.539	0.761	5.212	1.200
	Soil	CKB	6.920	1.817	1.382	0.761	4.808	1.225
	Soil	WEB	7.752	1.817	1.746	0.812	5.723	1.407
	Tx	CS	7.085	1.817	2.388	0.761	4.348	1.225
	Tx	NM	8.214	1.817	2.304	0.797	6.375	1.296
	Tx	RC	8.006	1.817	3.975	0.777	5.021	1.319
	Soil*Tx	ABC,CS	7.979	3.147	3.941	1.318	4.425	2.078
	Soil*Tx	ABC,NM	8.126	3.147	4.190	1.318	2.905	2.078
	Soil*Tx	ABC,RC	9.794	3.147	8.485	1.318	8.307	2.078
	Soil*Tx	CKB,CS	6.538	3.147	0.674	1.318	6.356	2.078
	Soil*Tx	CKB,NM	7.553	3.147	2.014	1.318	5.700	2.078
	Soil*Tx	CKB,RC	6.670	3.147	1.457	1.318	2.369	2.205
	Soil*Tx	WEB,CS	6.740	3.147	2.548	1.318	2.263	2.205
	Soil*Tx	WEB,NM	8.963	3.147	0.709	1.495	10.521	2.546
	Soil*Tx	WEB,RC	7.555	3.147	1.983	1.398	4.386	2.546
9/2/11	Soil	ABC	1.730	0.787	4.898	0.608	8.759	1.350
	Soil	CKB	3.699	0.787	2.877	0.608	5.830	1.378
	Soil	WEB	3.109	0.787	3.069	0.648	7.985	1.583
	Tx	CS	3.001	0.787	3.113	0.608	7.790	1.378
	Tx	NM	1.634	0.787	3.324	0.636	7.178	1.458
	Tx	RC	3.904	0.787	4.407	0.620	7.606	1.484
	Soil*Tx	ABC,CS	2.274	1.363	3.286	1.053	9.523	2.339
	Soil*Tx	ABC,NM	0.536	1.363	4.182	1.053	3.295	2.339
	Soil*Tx	ABC,RC	2.381	1.363	7.225	1.053	13.457	2.339
	Soil*Tx	CKB,CS	3.814	1.363	2.204	1.053	7.892	2.339
	Soil*Tx	CKB,NM	2.127	1.363	4.198	1.053	6.570	2.339
	Soil*Tx	CKB,RC	5.156	1.363	2.229	1.053	3.029	2.481
	Soil*Tx	WEB,CS	2.914	1.363	3.849	1.053	5.953	2.481
	Soil*Tx	WEB,NM	2.237	1.363	1.593	1.194	11.670	2.864
	Soil*Tx	WEB,RC	4.177	1.363	3.766	1.117	6.332	2.864
10/7/11	Soil	ABC	0.871	0.619	4.143	0.589	7.944	1.355

Table B.1 Continued

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
	Soil	CKB	2.909	0.619	2.797	0.589	5.839	1.383
	Soil	WEB	3.126	0.619	2.847	0.628	7.844	1.589
	Tx	CS	2.744	0.619	2.737	0.589	7.507	1.383
	Tx	NM	1.667	0.619	3.033	0.616	7.087	1.463
	Tx	RC	2.496	0.619	4.018	0.601	7.033	1.489
	Soil*Tx	ABC,CS	1.901	1.073	2.315	1.020	8.703	2.347
	Soil*Tx	ABC,NM	0.712	1.073	4.317	1.020	3.331	2.347
	Soil*Tx	ABC,RC	0.000	1.073	5.798	1.020	11.799	2.347
	Soil*Tx	CKB,CS	3.040	1.073	2.080	1.020	8.020	2.347
	Soil*Tx	CKB,NM	1.993	1.073	3.875	1.020	6.521	2.347
	Soil*Tx	CKB,RC	3.695	1.073	2.436	1.020	2.976	2.489
	Soil*Tx	WEB,CS	3.291	1.073	3.814	1.020	5.798	2.489
	Soil*Tx	WEB,NM	2.295	1.073	0.908	1.157	11.409	2.874
	Soil*Tx	WEB,RC	3.791	1.073	3.818	1.082	6.325	2.874
10/29/11	Soil	ABC	4.374	1.265	7.192	2.888	8.797	1.290
	Soil	CKB	6.462	1.265	8.142	2.888	6.044	1.317
	Soil	WEB	7.721	1.265	3.227	3.333	6.216	1.513
	Tx	CS	6.939	1.265	3.873	2.887	7.764	1.317
	Tx	NM	4.603	1.265	8.671	3.022	5.552	1.394
	Tx	RC	7.014	1.265	6.641	2.947	7.741	1.418
	Soil*Tx	ABC,CS	6.212	2.191	5.165	5.002	9.089	2.235
	Soil*Tx	ABC,NM	3.079	2.191	4.459	5.002	3.562	2.235
	Soil*Tx	ABC,RC	3.830	2.191	11.951	5.002	13.741	2.235
	Soil*Tx	CKB,CS	6.555	2.191	2.244	5.002	8.162	2.235
	Soil*Tx	CKB,NM	4.970	2.191	18.338	5.002	6.734	2.235
	Soil*Tx	CKB,RC	7.860	2.191	3.844	5.002	3.234	2.371
	Soil*Tx	WEB,CS	8.048	2.191	4.208	5.002	6.041	2.371
	Soil*Tx	WEB,NM	5.761	2.191	1.657	5.671	6.360	2.737
	Soil*Tx	WEB,RC	9.353	2.191	3.814	5.305	6.248	2.737
1/9/12	Soil	ABC	9.813	1.923	7.217	0.878	7.515	1.323
	Soil	CKB	10.18	1.923	2.840	0.878	4.763	1.351
	Soil	WEB	14.86	1.923	2.902	0.936	6.951	1.552
	Tx	CS	12.10	1.923	3.876	0.878	6.387	1.351
	Tx	NM	10.44	1.923	2.878	0.919	6.244	1.429
	Tx	RC	12.30	1.923	6.205	0.896	6.598	1.455
	Soil*Tx	ABC,CS	11.02	3.331	5.168	1.520	7.696	2.292

Table B.1 Continued

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
	Soil*Tx	ABC,NM	9.499	3.331	4.461	1.520	2.790	2.292
	Soil*Tx	ABC,RC	8.919	3.331	12.022	1.520	12.059	2.292
	Soil*Tx	CKB,CS	9.289	3.331	1.709	1.520	6.487	2.292
	Soil*Tx	CKB,NM	6.967	3.331	3.157	1.520	5.320	2.292
	Soil*Tx	CKB,RC	14.28	3.331	3.655	1.520	2.482	2.431
	Soil*Tx	WEB,CS	16.00	3.331	4.751	1.520	4.977	2.431
	Soil*Tx	WEB,NM	14.86	3.331	1.016	1.724	10.623	2.807
	Soil*Tx	WEB,RC	13.71	3.331	2.939	1.613	5.253	2.807
2/4/2012	Soil	ABC	9.902	2.171	8.229	1.067	9.562	1.583
	Soil	CKB	6.666	2.171	3.337	1.067	5.079	1.583
	Soil	WEB	11.94	2.171	4.205	1.185	9.255	2.085
	Tx	CS	9.592	2.171	5.357	1.067	8.029	1.635
	Tx	NM	9.519	2.171	4.558	1.152	7.757	1.828
	Tx	RC	9.397	2.171	5.856	1.102	8.111	1.828
	Soil*Tx	ABC,CS	7.978	3.761	7.813	1.848	11.694	2.742
	Soil*Tx	ABC,NM	11.06	3.761	6.969	1.848	4.462	2.742
	Soil*Tx	ABC,RC	10.66	3.761	9.905	1.848	12.531	2.742
	Soil*Tx	CKB,CS	7.160	3.761	2.015	1.848	4.347	2.742
	Soil*Tx	CKB,NM	5.446	3.761	4.942	1.848	7.093	2.742
	Soil*Tx	CKB,RC	7.390	3.761	3.054	1.848	3.798	2.742
	Soil*Tx	WEB,CS	13.63	3.761	6.242	1.848	8.045	3.004
	Soil*Tx	WEB,NM	12.04	3.761	1.763	2.263	11.717	3.878
	Soil*Tx	WEB,RC	10.14	3.761	4.609	2.024	8.004	3.878
3/3/2012	Soil	ABC	5.921	1.527	5.725	0.772	6.601	1.348
	Soil	CKB	7.082	1.527	2.199	0.772	4.594	1.376
	Soil	WEB	11.00	1.527	3.228	0.823	6.340	1.581
	Tx	CS	8.450	1.527	4.243	0.772	6.055	1.376
	Tx	NM	7.275	1.527	1.205	0.808	5.552	1.456
	Tx	RC	8.278	1.527	5.704	0.788	5.928	1.482
	Soil*Tx	ABC,CS	6.097	2.645	4.572	1.337	6.826	2.335
	Soil*Tx	ABC,NM	5.857	2.645	1.080	1.337	2.224	2.335
	Soil*Tx	ABC,RC	5.811	2.645	11.522	1.337	10.751	2.335
	Soil*Tx	CKB,CS	5.227	2.645	1.734	1.337	6.751	2.335
	Soil*Tx	CKB,NM	6.046	2.645	1.888	1.337	4.890	2.335
	Soil*Tx	CKB,RC	9.974	2.645	2.974	1.337	2.141	2.476
	Soil*Tx	WEB,CS	14.02	2.645	6.421	1.337	4.589	2.476

Table B.1 Continued

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
	Soil*Tx	WEB,NM	9.921	2.645	0.647	1.516	9.541	2.859
	Soil*Tx	WEB,RC	9.051	2.645	2.617	1.419	4.891	2.859
4/7/12	Soil	ABC	4.907	1.246	7.002	0.761	9.065	1.372
	Soil	CKB	9.876	1.246	4.725	0.761	5.072	1.400
	Soil	WEB	11.36	1.246	5.830	0.811	8.671	1.609
	Tx	CS	9.662	1.246	6.129	0.761	8.447	1.400
	Tx	NM	7.803	1.246	4.618	0.796	6.533	1.482
	Tx	RC	8.686	1.246	6.810	0.776	7.828	1.508
	Soil*Tx	ABC,CS	4.137	2.158	4.528	1.318	9.289	2.377
	Soil*Tx	ABC,NM	6.277	2.158	6.108	1.318	4.612	2.377
	Soil*Tx	ABC,RC	4.308	2.158	10.370	1.318	13.294	2.377
	Soil*Tx	CKB,CS	10.66	2.158	4.747	1.318	9.442	2.377
	Soil*Tx	CKB,NM	8.277	2.158	4.908	1.318	2.617	2.377
	Soil*Tx	CKB,RC	10.68	2.158	4.521	1.318	3.157	2.521
	Soil*Tx	WEB,CS	14.18	2.158	9.112	1.318	6.611	2.521
	Soil*Tx	WEB,NM	8.856	2.158	2.839	1.494	12.371	2.911
	Soil*Tx	WEB,RC	11.06	2.158	5.539	1.397	7.031	2.911
4/27/12	Soil	ABC	5.346	1.375	3.019	0.580	2.250	0.784
	Soil	CKB	4.007	1.375	1.748	0.580	2.619	0.800
	Soil	WEB	6.833	1.375	1.551	0.618	2.904	0.919
	Tx	CS	5.929	1.375	2.296	0.580	3.227	0.800
	Tx	NM	4.391	1.375	2.239	0.607	1.945	0.846
	Tx	RC	5.867	1.375	1.783	0.592	2.600	0.861
	Soil*Tx	ABC,CS	5.699	2.381	1.416	1.004	1.767	1.357
	Soil*Tx	ABC,NM	3.466	2.381	4.517	1.004	0.805	1.357
	Soil*Tx	ABC,RC	6.873	2.381	3.124	1.004	4.179	1.357
	Soil*Tx	CKB,CS	3.339	2.381	3.219	1.004	3.459	1.357
	Soil*Tx	CKB,NM	4.636	2.381	1.015	1.004	1.644	1.357
	Soil*Tx	CKB,RC	4.047	2.381	1.011	1.004	2.754	1.440
	Soil*Tx	WEB,CS	8.748	2.381	2.253	1.004	4.457	1.440
	Soil*Tx	WEB,NM	5.070	2.381	1.184	1.139	3.388	1.662
	Soil*Tx	WEB,RC	6.682	2.381	1.215	1.065	0.868	1.662
6/1/12	Soil	ABC	2.763	1.040	14.408	3.824	8.656	1.342
	Soil	CKB	6.421	1.040	4.531	3.824	6.232	1.370
	Soil	WEB	7.743	1.040	5.220	4.078	8.506	1.574
	Tx	CS	6.273	1.040	14.627	3.824	8.241	1.370

Table B.1 Continued

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
	Tx	NM	4.509	1.040	4.134	4.002	7.520	1.450
	Tx	RC	6.145	1.040	5.399	3.903	7.632	1.475
	Soil*Tx	ABC,CS	2.383	1.801	31.530	6.624	8.849	2.325
	Soil*Tx	ABC,NM	3.270	1.801	5.513	6.624	4.268	2.325
	Soil*Tx	ABC,RC	2.635	1.801	6.182	6.624	12.851	2.325
	Soil*Tx	CKB,CS	6.690	1.801	4.817	6.624	9.425	2.325
	Soil*Tx	CKB,NM	5.406	1.801	4.637	6.624	6.314	2.325
	Soil*Tx	CKB,RC	7.166	1.801	4.139	6.624	2.956	2.466
	Soil*Tx	WEB,CS	9.747	1.801	7.533	6.624	6.448	2.466
	Soil*Tx	WEB,NM	4.849	1.801	2.251	7.511	11.979	2.847
	Soil*Tx	WEB,RC	8.634	1.801	5.877	7.026	7.089	2.847
7/4/12	Soil	ABC	1.259	0.629	4.085	0.565	8.656	1.354
	Soil	CKB	2.428	0.629	3.569	0.565	6.625	1.382
	Soil	WEB	2.536	0.629	3.825	0.603	8.547	1.588
	Tx	CS	1.955	0.629	3.276	0.565	8.376	1.382
	Tx	NM	1.173	0.629	3.988	0.592	7.808	1.462
	Tx	RC	3.094	0.629	4.215	0.577	7.644	1.488
	Soil*Tx	ABC,CS	1.368	1.089	1.864	0.979	9.325	2.345
	Soil*Tx	ABC,NM	0.198	1.089	5.156	0.979	3.999	2.345
	Soil*Tx	ABC,RC	2.210	1.089	5.235	0.979	12.645	2.345
	Soil*Tx	CKB,CS	1.910	1.089	2.829	0.979	9.233	2.345
	Soil*Tx	CKB,NM	2.341	1.089	5.021	0.979	7.286	2.345
	Soil*Tx	CKB,RC	3.032	1.089	2.858	0.979	3.355	2.487
	Soil*Tx	WEB,CS	2.587	1.089	5.136	0.979	6.569	2.487
	Soil*Tx	WEB,NM	0.980	1.089	1.787	1.111	12.140	2.872
	Soil*Tx	WEB,RC	4.040	1.089	4.552	1.039	6.932	2.872
7/27/12	Soil	ABC	1.373	0.652	3.682	0.563	8.091	1.350
	Soil	CKB	2.277	0.652	3.218	0.563	6.340	1.378
	Soil	WEB	2.039	0.652	3.575	0.600	8.403	1.583
	Tx	CS	1.792	0.652	2.793	0.563	7.906	1.378
	Tx	NM	1.243	0.652	3.847	0.589	7.590	1.459
	Tx	RC	2.654	0.652	3.835	0.574	7.338	1.484
	Soil*Tx	ABC,CS	1.497	1.129	1.619	0.975	8.607	2.339
	Soil*Tx	ABC,NM	0.226	1.129	4.697	0.975	3.316	2.339
	Soil*Tx	ABC,RC	2.396	1.129	4.730	0.975	12.350	2.339
	Soil*Tx	CKB,CS	1.838	1.129	2.369	0.975	8.703	2.339

Table B.1 Continued

Date	Effect	Level	LS Mean	Std Error	LS Mean2	Std Error3	LS Mean	Std Error5
			0-60 (cm)		60-120 (cm)		120-180 (cm)	
	Soil*Tx	CKB,NM	2.692	1.129	4.839	0.975	7.210	2.339
	Soil*Tx	CKB,RC	2.301	1.129	2.446	0.975	3.107	2.481
	Soil*Tx	WEB,CS	2.041	1.129	4.391	0.975	6.408	2.481
	Soil*Tx	WEB,NM	0.812	1.129	2.005	1.105	12.244	2.865
	Soil*Tx	WEB,RC	3.266	1.129	4.329	1.034	6.557	2.865
9/7/12	Soil	ABC	0.741	0.466	3.494	0.581	8.098	1.259
	Soil	CKB	0.645	0.466	3.002	0.581	5.276	1.285
	Soil	WEB	0.718	0.466	3.073	0.619	8.114	1.477
	Tx	CS	0.314	0.466	2.268	0.581	6.262	1.285
	Tx	NM	0.645	0.466	3.563	0.608	7.360	1.360
	Tx	RC	1.146	0.466	3.739	0.593	7.867	1.384
	Soil*Tx	ABC,CS	0.073	0.807	1.172	1.006	8.621	2.181
	Soil*Tx	ABC,NM	0.000	0.807	4.497	1.006	3.237	2.181
	Soil*Tx	ABC,RC	2.149	0.807	4.814	1.006	12.437	2.181
	Soil*Tx	CKB,CS	0.121	0.807	1.585	1.006	3.876	2.181
	Soil*Tx	CKB,NM	1.542	0.807	4.841	1.006	6.946	2.181
	Soil*Tx	CKB,RC	0.272	0.807	2.582	1.006	5.005	2.314
	Soil*Tx	WEB,CS	0.746	0.807	4.047	1.006	6.287	2.314
	Soil*Tx	WEB,NM	0.392	0.807	1.352	1.140	11.896	2.672
	Soil*Tx	WEB,RC	1.016	0.807	3.820	1.067	6.159	2.672

APPENDIX C

SOIL WATER GRAPHS

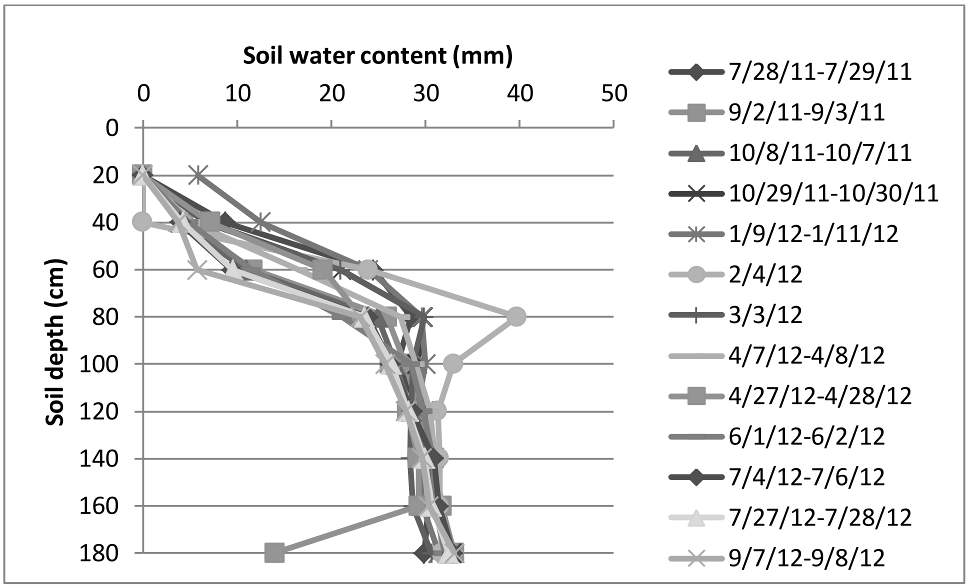

Figure C. 1. Soil water content by depth in sandy soil cut stump treatment.

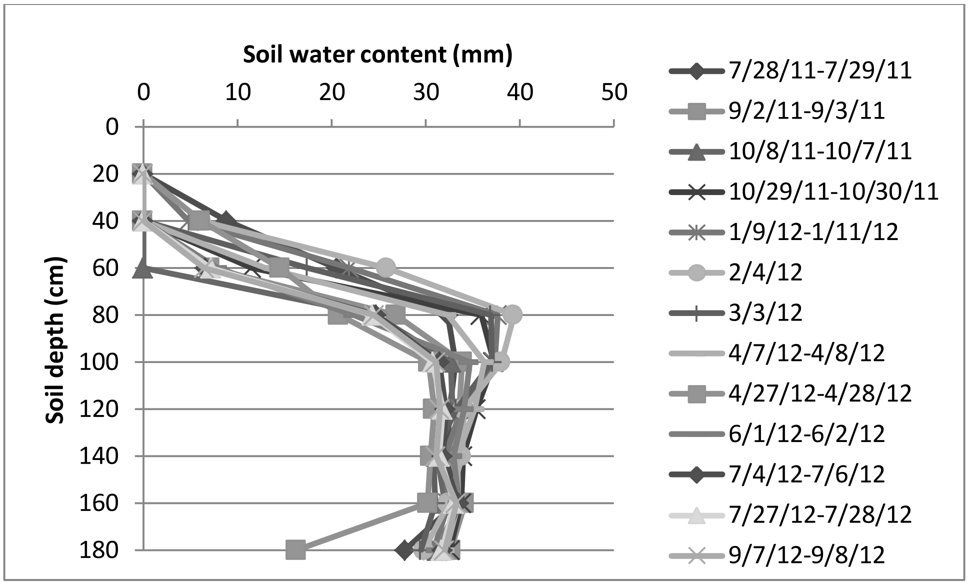

Figure C. 2. Soil water content by depth in sandy soil roller chop treatment.

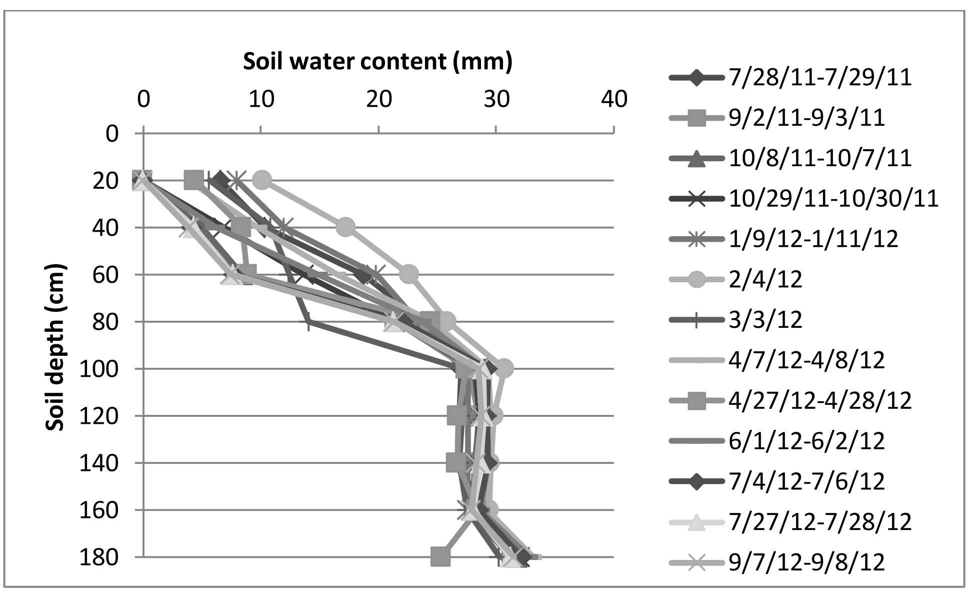

Figure C. 3. Soil water content by depth in sandy soil control treatment.

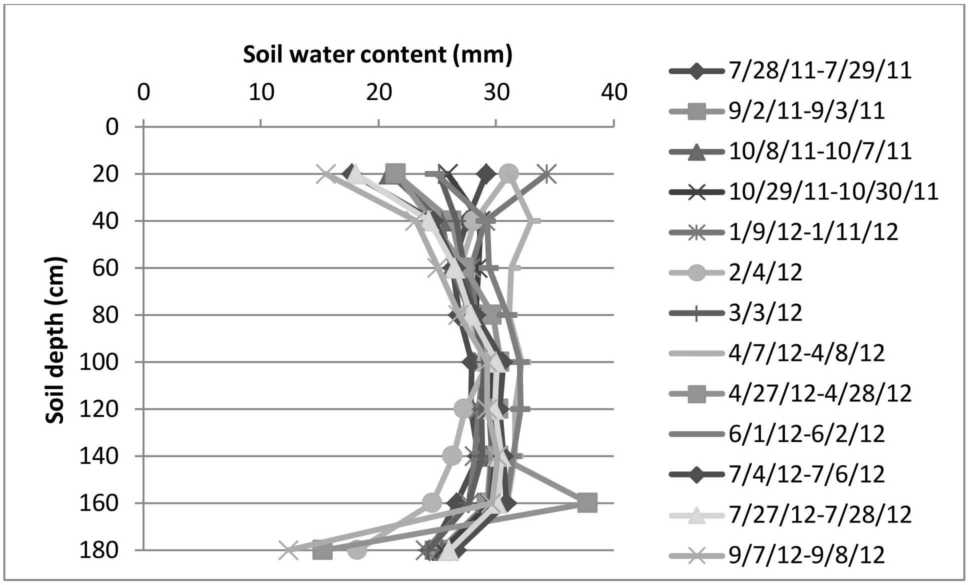

Figure C. 4. Soil water content by depth in clay loam soil cut stump treatment.

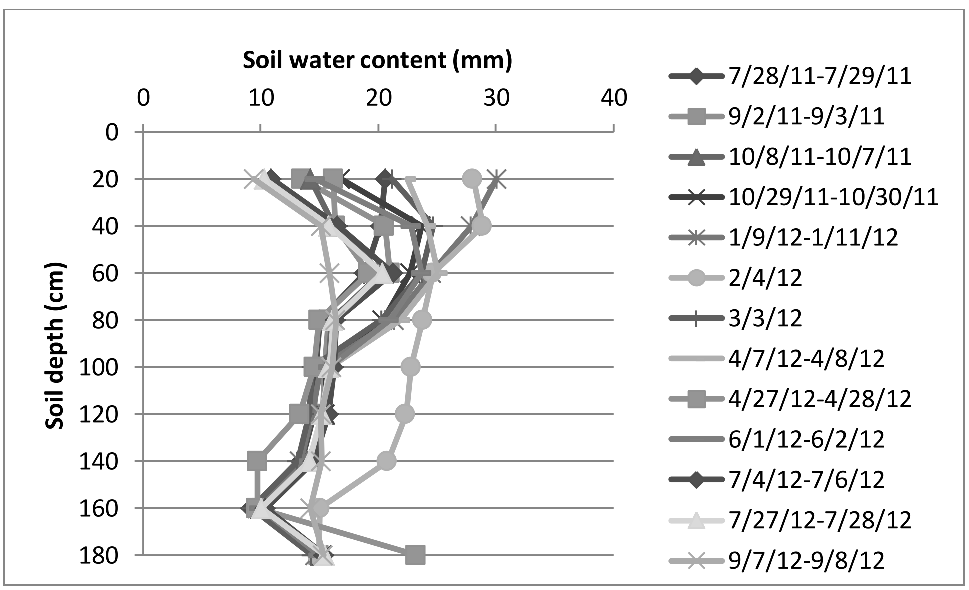

Figure C. 5. Soil water content by depth in clay loam soil roller chop treatment.

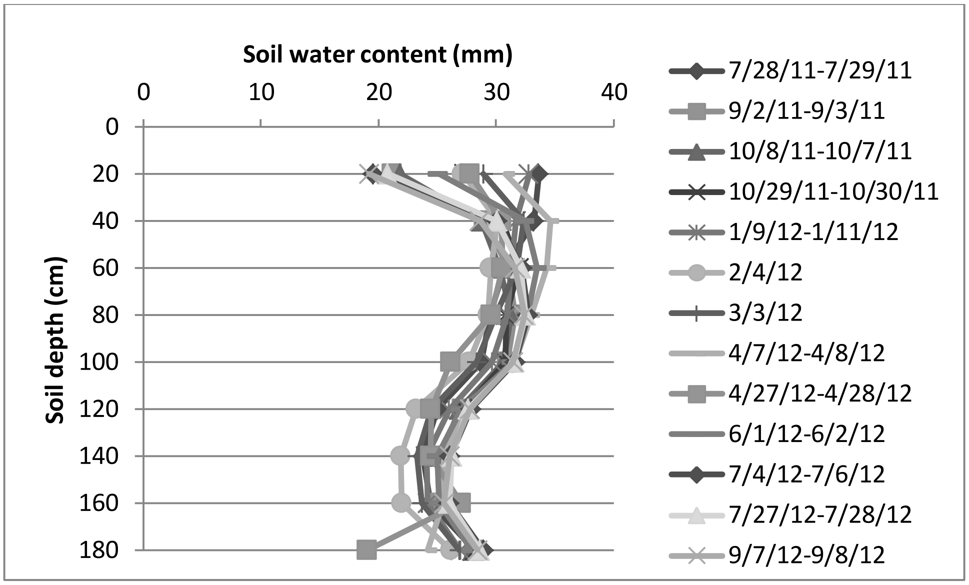

Figure C. 6. Soil water content by depth in clay loam soil control treatment.

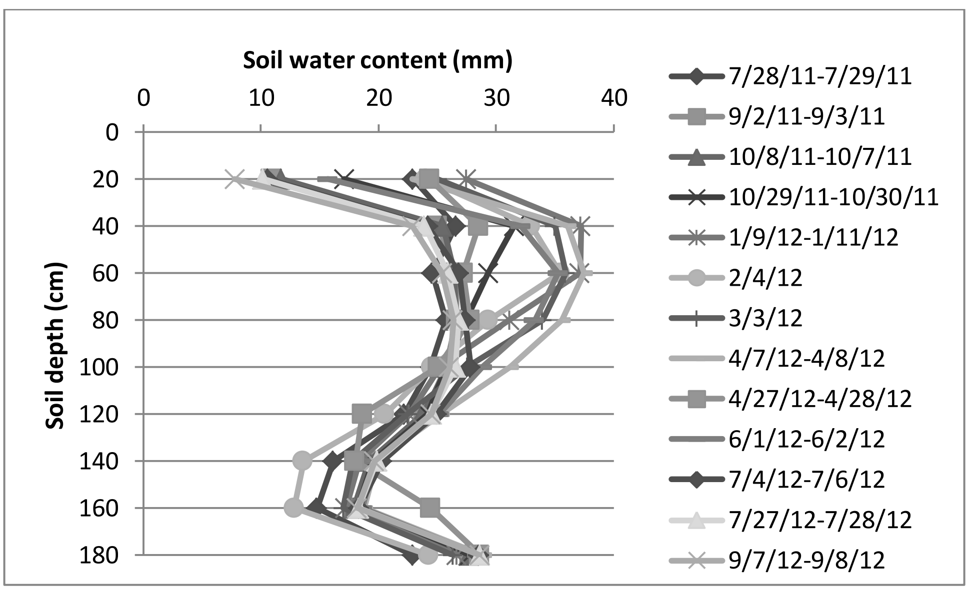

Figure C. 7. Soil water content by depth in loam soil cut stump treatment.

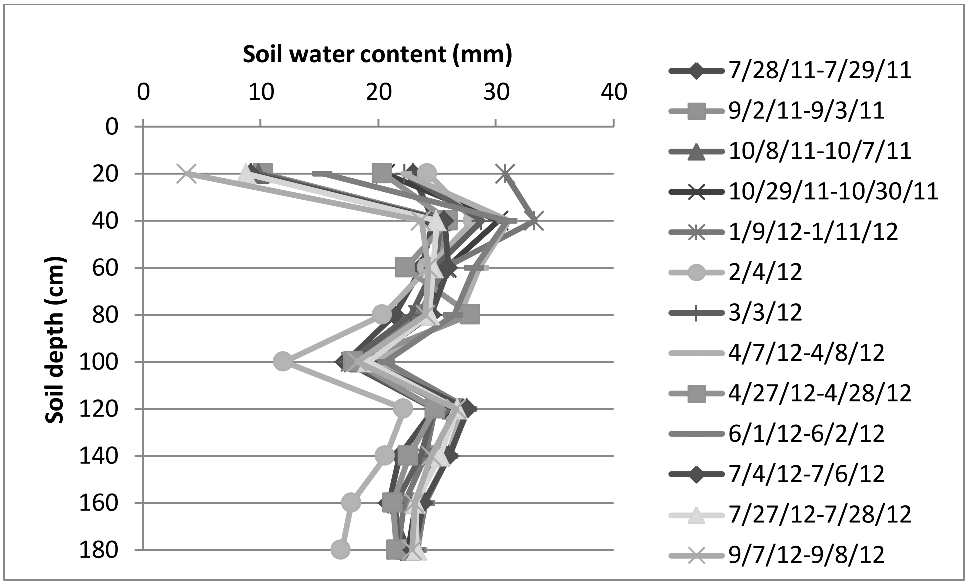

Figure C. 8. Soil water content by depth in loam soil roller chop treatment.

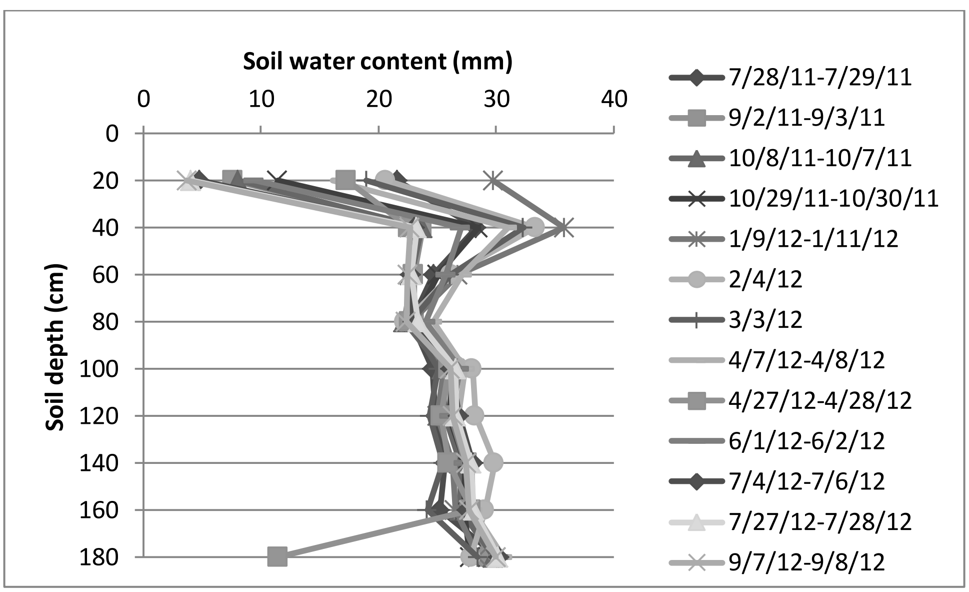

Figure C. 9. Soil water content by depth in loam soil control treatment.